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# Application of Multimodel Approach to Assessing the Stock Status of Anadromous Dolly Varden (*Salvelinus malma malma*) from the Rat River, Northwest Territories, Canada

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**Foreword**

This series documents the scientific basis for the evaluation of aquatic resources and ecosystems in Canada. As such, it addresses the issues of the day in the time frames required and the documents it contains are not intended as definitive statements on the subjects addressed but rather as progress reports on ongoing investigations.

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## TABLE OF CONTENTS

## ABSTRACT

The northern form of anadromous Dolly Varden (*Salvelinus malma malma*) from the Rat River has sustained an important subsistence fishery for Gwich'in and Inuvialuit communities situated in the Mackenzie Delta, Northwest Territories, Canada. During the late 1970s and early 1980s, declines in catches and the size of fish captured in the subsistence fishery prompted a concern that the population was experiencing reductions in population abundance. Consequently, this prompted the development of an annual community-based fisheries-dependent monitoring program for Rat River Dolly Varden. Implemented in its current form in 1995, the time series of subsistence catch and biological information were collected from three long term fixed locations during the seasonal return migration. Information on aquatic environmental conditions (water level, turbidity, debris content, and temperature), fishing characteristics (mesh size, length of the gillnets and fishing duration), and catch per unit effort (CPUE) was collected. In this study, we applied generalized linear mixed models and zero-augmented models to standardize the CPUE time series (1996–2014) associated with environmental variables and fishing behaviour and optimize quantitative model-derived estimates of population dynamics parameters. Multimodel inference indicated the best model for CPUE standardization was the zero-inflated Hurdle. Using the standardized CPUE time series, subsistence harvest statistics, and biological data, we structured three stock assessment models: depletion-based stock reduction analysis, surplus production model, and integrated statistical catch-at-age. Applying the model parameter weighting method, we assessed the optimal values (median ± standard deviation) of MSY (maximum sustainable yield), $N_{MSY}$ (population abundance at MSY) and $F_{MSY}$ (fishing mortality rate at MSY) for Rat River Dolly Varden between 1995–2014 were 1,301 ± 188 fish, 10,813 ± 1,555 fish, and 0.18 ± 0.02 per year, respectively. Our results indicated that during the late 1990s, the Rat River Dolly Varden stock was likely over-harvested, but since then the voluntary reduction of subsistence catches has optimistically benefited the gradual recovery of the anadromous population abundance in the western Arctic.

# INTRODUCTION

Quantitative models are effective at integrating various sources of fisheries information to collectively characterize stock status and its association with related sources of uncertainty (Hilborn and Walters 1992). There are approximately three groups of mathematical models conventionally applied to fisheries stock assessments. For example, depletion-based stock reduction analysis (DB-SRA, Dick and MacCall 2011) is a data-limited model which can utilize depletion-corrected average catch (DCAC, MacCall 2009) and stock reduction analysis (SRA, Walters et al. 2006) when harvest and general demographic parameters of the harvested population are the only available source of information. Deterministic SRA models use harvest data for the single stock trajectory while the stochastic SRA models apply the harvest data to explore the probability distributions of the fish stock size under alternative hypotheses about stable unfished recruitment rates and variability of assumed stock-recruitment relationships (Walters et al. 2006, Dick and MacCall 2011). Surplus production models (SPM, Schaefer 1957) have been used for the assessment of fisheries when a time series of abundance indices and harvest data are available. Because of its simplicity and relatively less data requirements, SPMs express the overall effects of growth, recruitment, and natural mortality by a comprehensive surplus component. The dynamic behavior of the biomass of the exploited fish populations can then be evaluated by differences between the surplus production and harvest removals (Hilborn and Walters 1992, Quinn and Deriso 1999). This process can be applied to generate biological reference points for management purposes (Caddy and Mahon 1995, Haddon 2001). As the complexity and extent of fisheries information for a stock increase, the more complicated and robust models, such as statistical catch-at-age models (SCA), are favoured by researchers and decision-makers for stock assessment (Hilborn and Walters 1992, Quinn and Deriso 1999). The underlying assumptions in SCA include constant year and age effects on the relationships between age-specific fishing mortality rates versus time-varying population abundance (Quinn and Deriso 1999). SCA models use a time series of total catch (in weight), abundance index, and catch-at-age observations along with some information on weight, maturity, and natural mortality at age to project the developmental histories of population status, exploitation, and spawner-recruitment as well as management advice. Overall, the selection of specific sets of mathematical or statistical models for fisheries stock assessment is largely determined by the nature and extent of the components of the fishery which can include uncertainties associated with the data collection (observation errors) and model structures (process errors), which can include traditional ecological knowledge (TEK) (Papik et al. 2003, Houde 2007, Roux et al. 2019).

Rather than using a single model for fish stock assessment, robust inferences can be made from several candidate models to maximize the uses of valuable information sources. Multimodel inference (MMI, Burnham and Anderson 2002) is a model selection framework that has recently gained some popularity as an alternative to null hypothesis significance testing. The MMI approach, using model selection and model averaging (Burnham and Anderson 2002), has been advocated as an alternative to null hypothesis significance testing. In particular, MMI can provide a means to simultaneously compare multiple model outputs, evaluate how well each is supported by data, and identify the best model(s). When model selection indicates no clear ‘best’ model(s), model averaging is then used to account for parameter uncertainty by calculating likelihoods among multiple competing hypotheses. These can be implemented in both information-theoretic and Bayesian frameworks (Burnham and Anderson 2002). In fisheries science, MMI has been applied to somatic growth patterns (Katsanevakis 2006, Katsanevakis and Maravelias 2008, Zhu et al. 2017), standardization of catch per unit effort (CPUE) of bycatch species (Brodziak and Walsh 2013), and stock assessment of Arctic Char from the Hornaday River (Zhu et al. 2017).

The northern form of anadromous Dolly Varden, *Salvelinus malma malma* (Walbaum, 1792), is geographically distributed west of the Mackenzie Delta in the Northwest Territories and along the Yukon and Alaska North Slope and south to the Aleutian Islands (Taylor et al. 2001). Anadromous Dolly Varden rear in natal drainages for approximately three to five years before undertaking seasonal freshwater-coastal marine migrations. Anadromous forms of Canadian Dolly Varden migrate to the Beaufort Sea in spring, approximately mid-June, to feed on marine prey (COSEWIC 2010). They are highly philopatric (Harris et al. 2015) and the return migration to their natal rivers typically begins in late July or early August.

The Rat River Dolly Varden is important for the subsistence and culture of Gwich'in and Inuvialuit people from the communities of Aklavik and Ft. McPherson, Northwest Territoires (NT) (Papik et al. 2003, Sandstrom et al. 2009, Byers et al. 2019). During the late 1970s and early 1980s, declines in catches and the body size of the Rat River Dolly Varden prompted concern that the population was experiencing a decrease in abundance. Consequently, this prompted the establishment of a community-based fisheries-dependent monitoring program for Rat River Dolly Varden in 1989 (Harwood 2001, Harwood et al. 2009) that expanded to multiple sampling sites in 1995 (Roux et al. 2012, Gallagher et al. 2020).

Despite the importance of the Rat River Dolly Varden in the western Canadian Arctic, there have been no quantitative stock assessment models to characterize population dynamics and evaluate the stock status. In this study, we integrated multiple datasets from the community-based monitoring program to estimate the population dynamics parameters and elucidate the temporal development of the stock status. Our objectives were to: 1) summarize the demographic parameters of the stock, 2) standardize CPUE time series concerning variable environmental conditions and fishing behaviors, and 3) formulate population dynamics parameters relevant to future management options. Overall, this study will enhance our understanding of fish population dynamics for Rat River Dolly Varden and consolidate operational considerations of optimum model-based management strategies.

# MATERIALS AND METHODS

## STUDY AREA

The headwaters of the Rat River in the Richardson Mountains straddle the border of the Yukon and Northwest Territories. The river flows along a 130 km course and drains into the Husky Channel of the Mackenzie Delta, 35 km northwest of Fort McPherson, NT (Harwood et al. 2009). The Rat River watershed drains an area of 1,680 km$^2$ (Gillman and Sparling 1985) and the topography consists of rugged mountains in the upper reaches, rolling foothills in the central portion, and low-lying spruce muskeg near the mouth (Gillman and Sparling 1985). The high gradient rivers in this area freeze completely and cease flowing in winter apart from locations where perennial groundwater maintains sections of open water year round. Locations of perennial groundwater are associated with the Dolly Varden spawning and overwintering habitat (Mochnacz et al. 2010).

## DATA SOURCES

Several datasets were used in this study: 1) subsistence harvest from 1986–2014 (Lea et al. 2021); 2) biological data from several exploratory and monitoring programs which is synchronized with documentation of subsistence harvest during 1986–2014. Through implementation of these programs, a set of biological information, including fork length, round weight, sex and maturity, was gathered using various gear types like gillnet, angling, electroshock, seine, and hoopnet (Gallagher et al. 2020), and 3) catch-per-unit effort (CPUE) index from the community-based Rat River Dolly Varden monitoring program in the period 1996–2014 (Gallagher et al. 2013).

The community-based Rat River Dolly Varden monitoring program started in 1995 and was fully implemented in 1996 at three traditional fishing camps 1) Big Eddy (BE; 67°58' N, 135°20' W) in the Husky Channel of the Mackenzie Delta, 2) the mouth of the Rat River (RR; 67°45' N, 135°8' W), and 3) a location called Destruction City (DC; 67°45' N, 135°23'W) in the Rat River proper (Figure 1). During 1996–2014, monitors collected the following for each fishing set: 1) subsistence harvest (number of Dolly Varden caught); 2) environmental conditions (water temperature, level, colour, turbidity, and debris concentration); 3) fishing effort (net number and length, mesh size and net soak time), 4) biological data from ~ 200 randomly-selected fish samples taken from the harvest at each location (Table 1). Biological data included fork length, round weight, sex, maturity, gonad weight, and otoliths of each sampled fish. The methods, criteria, and protocol for the age determination of the fish are referred to in Gallagher et al. (2020). Air temperature was downloaded from the National Weather Network portal.

## CPUE STANDARDIZATION

### Data exploration

We followed the underlying assumption of Gaussian distribution to examine the outliers, heterogeneity of variance, collinearity, and zero tolerance (Zuur et al. 2010, Hilbe 2011) in the Rat River Dolly Varden subsistence fishery monitoring dataset(s), for both the response (CPUE) and explanatory (environmental and fishing effort) variables (Table 1). Here, outliers are observations with relatively large or small values compared to most observations that cause over-dispersion in a Poisson or other GLMs (Venables and Ripley 2002, Zuur et al. 2010). Using analysis of variance (ANOVA) and regression-related models, heterogeneity of variance can be detected in cases where the ratio between the largest and smallest variance is ≥ 4, which could negatively affect least-square estimators (Zar 2010). Collinearity or a significant correlation between explanatory covariates (Zuur et al. 2010) is assumed when correlation coefficients for covariates are > 0.8 (McCullagh and Nelder 1989, Zar 2010). Because unbalanced design results from this type of stratified data (Zar 2010), we explored two Gaussian-based statistical analyses: a fixed-effect ANOVA and count regression modeling to remove redundant or collinear variables from the dataset.

Three types of sums of squares (SS) ANOVA were used to detect collinearity from excessive effects of covariates (McCullagh and Nelder 1989) and determine the "best" subset of explanatory variables for the standardization of the CPUE dataset. Type one SS analysis adds individual variables sequentially to test how the different combinations of variables impact the model results. In type two SS analysis, each effect is adjusted for all other terms except ones that "contain" the effect being tested. Therefore, type two SS analysis does not use constraints on the parameters and alternative model specifications can produce identical results (McCullagh and Nelder 1989). In type three SS analysis, the SS would be obtained for each variable when entered last into the model. The effect of each variable is evaluated after all other factors have been accounted for. Type three SS does not test hypotheses about least squares means. Instead, it tests hypotheses that are complex functions of the patterns of missing cells in higher-order interactions and that are typically not meaningful.

### CPUE models

During the community-based monitoring period (1996–2014), nominal values of CPUE were determined by summing the total catch from 22.86 m (25 yds long) gillnets, set in water for 24 hrs. There can be considerable uncertainty in CPUE data, mainly due to variation in local hydroclimate conditions, changes in gear structure and fishing behaviour, as well as the nature of seasonal migration behaviours under investigation. The start timing, peak, and duration of down- or upstream migrations of anadromous Dolly Varden are essentially impacted by hydroclimate conditions and thermal variables (Jensen et al. 2020). Consequently, zero catch has become rather common in the community-based fishery-dependent monitoring activity.

Zero catch poses a real challenge in fisheries science given that these cases could either indicate the absence of fish or poor fishing gear performance (Martin et al. 2005). If the extent of zero values is so great that the dataset does not readily fit into a standard distribution (e.g., normal, Poisson, binomial, negative-binomial, and beta), the alternative could be to use a zero-inflated approach (Heilbron 1994). Simply excluding or replacing the zero values with the average will result in the loss of important information about the capture efficiency of a specific fishing gear type (Arab et al. 2008, Brooks et al. 2017). Therefore, standardization of CPUE is necessary to account for the non-normal distribution of the count data (Maunder and Punt 2004).

## CPUE model analysis and multimodel inference

We used three generalized linear models (GLMs) (Poisson, Quasi-Poisson, and negative binomial regression models) and three zero augmented models (ZAMs) (Hurdle, zero-inflated Poisson (ZIP), and zero-inflated negative binomial (ZINB)) to standardize the CPUE for Rat River Dolly Varden (Table 2). Among GLMs, the Poisson regression model assumes the probability of the parameter distribution follows a constant in time or space at each sampling unit while quasi-likelihood Poisson (Wedderburn 1974) and negative binomial (Potts and Elith 2006, Lindén and Mäntyniemi 2011, Brodziak and Walsh 2013) can be applied for over-dispersed count data where the conditional variance exceeds the conditional mean. ZAMs, also termed two-component mixture models, combine a point mass at zero with a proper count distribution function (Potts and Elith 2006, Zeileis et al. 2008, Brodziak and Walsh 2013). Hurdle models consist of two parts that combine a truncated component for positive counts and a hurdle component for zero counts that permits a stochastic process (Cragg 1971, Martin et al. 2005, Zeileis et al. 2008). A logit link function was applied to ZIP and ZINB whose starting values were estimated by the expectation-maximization algorithm.

Data were read and graphed using gdata, foreign and ggplot. Pair-wise comparison by scatter plots was used to identify possible collinearity between explanatory variables such as spatiotemporal dynamics, aquatic environment, and fishing behaviour. When significant correlations were identified, variance inflation factors (VIFs) were used to detect possible collinearity between variables (Cattelan 2012). Two selection criteria, $r$ = 0.80 (Zar 2010) and VIF < 3 (Zuur et al. 2010), were applied to test the presence of collinearity. ANOVA, GLMs, and ZAMs were performed using several statistical packages under R operational environment (R Core Team 2023). Statistical analyses were performed using Hmisc for pairwise correlation, Companion to Applied Regression (CAR) for types I, II, and III sums of squares, leaps for stepwise regression, AER for regression analysis of count data, and pscl for ZAM. Analysis of covariance (ANCOVA) was used to test the differences in slopes and intercepts among log-transformed length and weight regressions. The intercepts and slopes reflect differences in magnitude and rate, respectively. The statistical significance level was set at $\alpha = 0.05$. Growth parameters for VBGM were estimated using ADMB.

Multimodel inference (MMI) is a model selection framework that has recently gained popularity as an alternative to null hypothesis significance testing. Here, we used MMI to examine the CPUE models using small-sample-size corrected Akaike information criterion (AICc) to rank all candidate models (Burnham and Anderson 2002). AICc differences and weight of evidence ($w_i$) can be computed to assign the relative importance of all working models according to how well these are supported by the data (Burnham and Anderson 2002). Then, single or multiple sets of candidate models can be selected in terms of AICc and weight of evidence ($w_i$) for standardization of CPUEs, see the formula in Table 2.

## BIOLOGICAL PARAMETERS

### Temporal trends in length, weight and proportion at age

Mean fork length (mm) and round weight (g) of fish samples collected from subsistence harvest were represented to the temporal trend of the size change during 1986–2014. The composition of fork length, round weight by box plots, and age composition by circle plots for fish sampled during the Rat River Dolly Varden monitoring program (1995–2014) was examined for temporal variation of fish population biology at each location (BE, DC and RR). An overall average was also calculated for the time period.

### Length-weight relationship

A power function was used to determine the length-weight relationship for Rat River Dolly Varden using data from all locations (BE, DC, RR and Fish Creek) and gear types (using gillnet, angling, electroshock, seine, and hoopnet), 1986–2014 (*n* = 13,549, Table 3):

$$W = \mathrm{a}L^{\mathrm{b}} \text{ or}$$

$$log(W) = log(\mathrm{a}) + \mathrm{b} \times log(L)$$

where, *L* = fork length (± 1 mm), *W* = round weight (± 1 g), and a and b are regression coefficients. Both coefficients have important applications in monitoring the growth and production status of fish populations under assumptions of a steady recruitment state and effective sample size (ESS). When b ≈ 3, weight increases are considered isometric, while increases are allometric when b is otherwise. Mean and standard deviation for length and weight parameters for this overall dataset were also estimated.

### Growth

Age, length and weight from 7,139 fish (2,535 male, 4,499 female and 105 unknown fish) sampled from BE, DC, RR and Fish Creek using gillnet, angling, electroshock, seine, and hoopnet between 1986–2014, were used for the fish size-at-age growth model analysis. The von Bertalanffy growth model (VBGM) is commonly used to describe the change in fork length ($L_t$: mm) or round weight ($W_t$: g) with age (Panik 2014):

$$L_t = L_\infty\left(1 - e^{-K(t-t_0)}\right)$$

$$W_t = W_\infty\left(1 - e^{-K(t-t_0)}\right)^{\mathrm{b}}$$

where $L_\infty$ and $W_\infty$ are asymptote length and weight when fish approaches age infinity, *K* and $t_0$ are Brody growth coefficient and assumed age when length approaches zero, respectively.

### Natural mortality

We chose three age-based models to estimate *M*: 1) the life history model (Chen and Watanabe 1989), 2) the body length scaling model (Gislason et al. 2010), and 3) the body mass scaling model (Peterson and Wroblewski 1984, Lorenzen 1996, 2000). The life history model incorporates parameters obtained from age-at-maturity and the VBGM (Chen and Watanabe 1989). Peterson and Wroblewski (1984) used a theoretical size-spectrum model to generate an exponent of -0.25 for *M* to scale the unit of weight. Similarly, Lorenzen (1996, 2000) modelled *M* using a power function of weight-mortality for a variety of freshwater and marine species. Lorenzen (1996) compared the estimated natural mortality of fishes from freshwater, marine, and aquaculture ponds and concluded that no significant differences were found among these ecosystems.

- Life history model (Chen and Watanabe 1989):

$$M_t = \begin{cases} \dfrac{K}{1-e^{-K(t-t_0)}}, t < t_m \\ \dfrac{K}{a_0 + a_1(t-t_m) + a_2(t-t_m)^2}, t \geq t_m \end{cases}$$

$$\mathrm{a}_0 = 1 - e^{-K(t_m - t_0)}$$

$$a_1 = Ke^{-K(t_m - t_0)}$$

$$a_2 = -0.5K^2 e^{-K(t_m - t_0)}$$

$$t_m = -\frac{1}{K}\ln|1 - e^{Kt_0}| + t_0$$

- Length-based model (Gislason et al. 2010):

$$ln(M_t) = 0.55 - 1.61ln(L_t) + 1.44ln(L_\infty) + ln(K)$$

- Weight-based model (Peterson and Wroblewski 1984; Lorenzen 1996, 2000):

$$M_t = 1.92(0.20W_t)^{-0.25}$$

$$M_t = 1.69{W_t}^{-0.292}$$

$L_\infty$, $K$ and $t_0$ are VBGM parameters described above, $M_t$ is natural mortality at age $t$ and $t_m$ is the estimated age when fish reaches maturity.

The geometric means of these three models were also calculated for males, females, and sexes combined.

## QUANTITATIVE FISHERY ASSESSMENT MODELS

### Depletion-based stock reduction analysis

To construct the Rat River Dolly Varden DB-SRA model (Table 4), we specified a group of probability distributions to generate the initial values of the biological parameters. It included 1) logistic distribution to delineate 50% age-at-maturity from the biological studies (Roux et al. 2012; Gallagher et al. 2020); 2) lognormal distribution for the mean value of age-dependent $M$ and 0.20 for log-scale standard deviation; 3) $F_{MSY}/M$ was lognormally distributed with an expectation of 0.80 and log-scale standard deviation of 0.20 (Walters and Martell 2004); 4) beta distribution was used to calculate the mean relative depletion level $B_T/K$, and was bounded by 0.01 and 0.99 (Dick and MacCall 2011). We chose a time period (1991–2005) when the subsistence harvest demonstrated strong high/low harvest contrast to calculate the initial DCAC. We ran a total of 100,000 Monte Carlo simulations to characterize the imprecision of fisheries parameters governing stock productivity and status.

### Surplus production model

The standardized CPUE (Hurdle model) and subsistence harvest data from 1996–2014 were used as inputs to a standard Schaefer-based SPM for Rat River Dolly Varden. We explored automatic differentiation model builder (ADMB software) and maximum likelihood estimates

(MLEs, Zar 2010) to derive a set of key population dynamics parameters (Table 5). To compare model performance, we conducted a Markov chain Monte Carlo (MCMC, Gelman et al. 2004) simulation with 5,000,000 iterations in which every 250 simulations was sampled, leading to one chain of 20,000 samples for all surplus production model parameters as defined in Table 5, including carrying capacity (*K*), intrinsic population growth rate (*r*), and catchability efficient (q). To reduce the confounding effects between biomass at year *t* ($B_t$) and *K* (Meyer and Millar 1999a, b; Millar and Meyer 2000), a discrete form of the SPM was re-parameterized by relative biomass ($P_t = B_t/K$) to express the annual biomass conditional to *K*.

## Integrated statistical catch-at-age model

We used a variation of SCA called integrated statistical catch-at-age model (iSCAM), with the following assumptions: 1) the length-at-age growth pattern followed the time-invariant VBGM (Hilborn and Walters 1992); 2) maturity-at-age parameters conformed to an asymptotic or logistic function (Quinn and Deriso 1999, Chuwen et al. 2011, Thorson and Prager 2011); 3) gillnet selectivity followed a bell-shaped function (Shoup and Ryswyk 2016); 4) recruitment and spawner relationship followed Beverton-Holt relationship (Beverton and Holt 1957); and, 5) subsistence harvest was reported without error (Roux et al. 2012, Gallagher et al. 2020, Lea et al. 2021). We were especially interested in using iSCAM to: 1) integrate community-based monitoring information into an assessment model; 2) construct a trend in abundance; and, 3) evaluate model performance (Table 6 and Table 7).

Subsistence harvest and biological data from the exploratory and monitoring programs conducted during 1986–2014 were used as sources for model inputs. CPUE was standardized using the Hurdle model. The information on the maturity-at-age was referred to Gallagher et al. (2020). Because the median day of the year (DOY) (Julian) for Rat River Dolly Varden harvest monitoring was August 14 (DOY = 226), the monitoring time in the model was set to 0.62 (226/365). To incorporate the selectivity of gillnet and catch-at-age information, the observed catch-at-age composition inputs were limited by means of 113 mm and 127 mm mesh-sized gillnets (Gallagher et al. 2020). Ages ranging from 3 to 13 yrs, and fish ≥ 10 yrs of age were lumped into a single age class (10+).

To obtain effective samples estimating the median values and credible intervals of 58 model parameters from the joint posterior distribution, we used the Metropolis-Hastings routine that is built into ADMB. After discarding the first 10,000 samples, we conducted MCMC simulations with a chain of length 2,500,000 and sampled every 250th simulation. Applying penalized MLE, the model parameters were estimated including $\widehat{N_1}$ the initial number-at-age present in the first year (e.g., 1996), $R_{\text{age-1}}$ (i.e., recruitment from 1996 to 2014), and $log\bar{R}$, which is the log-transformed average age-1 recruitment over the time period examined (i.e., 1996–2014). The details of the notation for the statistical catch-at-age stock assessment model and the state-space catch-at-age calculations are found in Table 6 and Table 7, respectively.

## Model averaging

Multimodel inference (MMI) is a model selection framework as an alternative to null hypothesis significance testing. This type of inference favours determining a single best “final” model parameter set. We used Monte Carlo simulation to assess the bias and mean squared error of the model estimates, weighting sample size and weighting by the relative variance of each effect size (Marín-Martínez and Sánchez-Meca 2010). Hartung et al. (2008) explored optimal weight for averaging a set of independent inverse variances (WIV) of each effective sample size to evaluate the applicability of model-derived management parameters. These multimodel averaging approaches are considered a form of MMI (Marín-Martínez and Sánchez-Meca 2010, Harrison 2011).

We compiled median estimates for maximum sustainable yield (MSY), abundance at MSY ($N_{MSY}$), fishing mortality at MSY ($F_{MSY}$), biomass at MSY ($B_{MSY}$), exploitation rate at MSY ($E_{MSY}$), and stock status indicators for exploitation ($F/F_{MSY}$), and biomass ($B/B_{MSY}$) from DB-SRA, SPM and iSCAM into a grid (Quinn and Deriso 1999). We then used WIV to aggregate MSY, $N_{MSY}$, $B_{MSY}$, $F_{MSY}$, and $E_{MSY}$, to minimize the variance of the weighted average. Each random variable is weighted in inverse proportion to its variance. We followed the recommendation by Marín-Martínez and Sánchez-Meca (2010) and adopted the average ($\hat{\mu}$) of k independent standardized mean differences:

$$\hat{\mu} = \frac{\sum_{i=1} \widehat{w_i} d_i}{\sum_{i=1} \widehat{w_i}}$$

where $w_i = \frac{1}{(\hat{\sigma}_i^2 + \hat{\tau}_i^2)}$ and $\hat{\sigma}_i^2 \, and \, \hat{\tau}_i^2$ are random effects within variance and between variance, respectively. Parameter $d_i$ is an unbiased estimator of the population's standardized mean differences between pairs of model parameters.

# RESULTS

## SUMMARY OF COMMUNITY-BASED FISHERY MONITORING (1995–2014)

### Monitoring period

From 1996 to 2014, the earliest and latest DOYs were recorded on July 23 (DOY 204) and September 24 (DOY 267). A majority (78%) of monitoring activities occurred from August 17 through September 6 (DOY 230–250), while 4% (178 sets) and 17% (660 sets) occurred from July 23 through August 7 (DOY 204–220) and September 7–24 (DOY 251–267), respectively. The monitoring period varied among years ($F_{18} = 65.30$, $p < 0.0001$), and locations ($F_2 = 192.69$, $p < 0.0001$) with a strong interaction between year and location ($F_{36} = 4.46$, $p < 0.0001$). More than 10% of monitoring sets were taken at BE before August 7 (DOY 220), compared to less than 1% in DC and 3% in RR (Figure 2). After September 6 (DOY 250), 9% (96 sets), 17% (197 sets), and 23% (367 sets) were in BE, RR, and DC, respectively. Monitoring activities occurred earliest at BE while monitoring duration was longest at DC.

### Environmental condition

Daily water level was predominantly recorded as 'low' (40% of net sets) followed by 'normal' (31% of net sets) and 'high' (29% of net sets). Water colour was mainly described as 'clear' (52% of net sets) followed by dirty (48% of net sets), while debris concentration was mostly 'lots' (52% of net sets) followed by 'few' (43% of net sets) and 'none' (5% of net sets) (Figure 3). Two-way ANOVA indicated water level varied significantly with year ($F_{18} = 54.00$, $p > 0.0001$) and location ($F_2 = 266.58$, $p < 0.0001$), and there was a significant interaction between these two variables ($F_{36} = 17.55$, $p < 0.0001$). Nets were set in 'low' water level conditions most frequently at DC (55%), followed by BE (31%) and RR (28%). 'High' water level conditions were most frequent for sets at RR (41%), followed by BE (38%) and DC (15%).

Water turbidity is associated with water colour (clear or dirty) and concentration of debris (none, some, or lots). Two-way ANOVA showed that both years ($F_{18} = 74.65$) and locations ($F_2 = 139.54$) explained a significant amount of variation in water turbidity ($p<0.0001$) along with a strong interaction between year and location ($F_{36} = 15.37$, $p < 0.0001$). A Tukey post-hoc test revealed significant pairwise differences in turbidity between RR and BE (1.01), between BE and DC (0.54), and between RR and DC (0.46) (Figure 3).

Water temperature was compared using a two-way ANOVA and post hoc multiple comparisons (Figure 4). Both year ($F_{18}$ = 106.69) and location ($F_2$ = 397.29) explained a significant amount of variation in water temperature (Figure 4). Over the entire monitoring period, temperature was warmer in 1996–1998 and 2010–2013, and lower than average during 2000–2008.

We performed pairwise comparison analysis to identify the collinearity between the explanatory variables year, DOY, water level, turbidity, and water temperature (Figure 5). Two pairs of bivariate collinearity were detected, a negative correlation between DOY and temperature ($r$ = -0.62, $p$ < 0.0001) as well as a positive correlation between water level and turbidity ($r$ = 0.56, $p$ < 0.0001) (Figure 5). We found collinearity between DOY and temperature with most (< 98%) occurring after DOY 217 (August 5) when daily temperatures began to shift from warm summer to a cooler fall. The VIFs of DOY and water temperature as well as turbidity and water level were > 1.7, therefore, we decided to remove the explanatory variables DOY and water level to avoid possible collinearity (Zuur et al. 2010), leaving year, turbidity, and water temperatures as the explanatory variable set.

## Changes in fishing effort and subsistence harvest

Between 1996 and 2014, a total of 3,862 gillnet sets were recorded at three long-term community-based monitoring locations from the Rat River Dolly Varden subsistence fishery. After removing 11 sets with either inaccurate date records or extreme values for soak times (i.e., < 1 hour or > 30 hours), the remaining 3,851 effective gillnet set records were retained to evaluate temporal variation in CPUE. The annual effective sets varied with the year ($F_{18}$ = 6.56, $p$ < 0.0001) and locations ($F_2$ = 12.90, $p$ < 0.0001, Figure 6). Among years, the proportion of total gillnet sets at BE, DC, and RR was 27%, 42%, and 31%, respectively. Among three locations, the minimum and maximum gillnet sets were 88 in 2008 and 488 in 2001, respectively.

In addition to set numbers, fishing effort information included data on mesh size, net panel length, and fishing duration. Two-way ANOVA revealed that both year ($F_{18}$ = 152.57) and location ($F_2$ = 318.82) explained a significant amount ($p$ < 0.0001) of variation in gillnet mesh size (Figure 7). At BE, the predominant mesh size was 102 mm (560 sets; 54%) compared to 114 mm (478 sets; 46%). Two sets of 140 mm mesh gill nets were used in 2005 only. Among 1,635 net sets at DC, the 102 mm mesh size was the most frequently used (1,339 sets; 82%) followed by 114 mm (218 sets; 13%), 127 mm (70 sets; 4%), and 89 mm (7 sets, < 1%). There were a total of 1,177 gillnet sets in the mouth of RR, 53% (618 sets) were 102 mm, 42% (491 sets) 114 mm, and 6% (68 sets) 127 mm.

ANOVA demonstrated that gillnet panel length, which ranged from 1 m to 30 m, varied significantly by year ($F_{18}$ = 221.0) and location ($F_2$ = 1513.1, Figure 8). At BE, between 1996 and 2004, most nets (> 77%) were 20–25 m in length. Then from 2005 to 2014 the majority (83%) of nets were 15–20 m. At DC, gillnet lengths of 15–20 m were most frequent (98%) until 2009 when lengths < 15 m were present. At RR, gillnet length was predominantly 20–25 m (42%), but 10.1–15 m (25%), 15.1–20 m (20%), and < 10 m (13%) were also used.

Fishing duration (i.e., hours soaked) was significantly different ($p$ < 0.0001) for both years ($F_{18}$ = 94.35) and locations ($F_2$ = 2,116.25, Figure 8). Fishing duration at BE and RR were quite similar, with ~ 50% and 71% of sets having a duration of 21–25 hrs and 27% and 22% with a duration of 10–15 hrs, respectively. Fishing duration at DC was considerably shorter than at BE or RR, with 10–15 hr sets dominating (62%), followed by 5–10 hr sets (33%).

The reported subsistence harvest of Rat River Dolly Varden peaked at 3,760 fish in 1998. Catch has declined since then, with only 387 fish reported in 2014 (Table 8 and Figure 9).

From 1998 to 2014 the average fork length (474.30 ± 69.77 mm) and round weight (1,272.70 ± 601.84 g) were 15% and 48% greater, respectively, than the corresponding average for length (411.70 ± 75.55 mm) and weight (858.06 ± 403.41 g) during 1986 to 1997 (Figure 9). This temporal variation in average body size was accounted for in the estimation of annual biomass from counts of individual fish through the application of the length-weight model estimations.

## MODEL INPUTS

### CPUE standardization

Nominal (unstandardized) CPUE was significantly different among years ($F_{18} = 68.21$, $p < 0.0001$) and locations ($F_2 = 4.31$, $p < 0.05$, Figures 10 and 11). The highest average CPUE was 8.60 ± 0.26 fish at BE, followed by 7.46 ± 0.24 fish at DC and 7.41 ± 0.42 fish at RR (Figure 10). The coefficient of variation in the nominal CPUE was lowest for BE (96%) followed by DC (132%) and RR (195%). Among all sets, the frequency of fish per set demonstrated an expected left-skewed distribution (Figure 11). Zero catch accounted for 27% of the entire data set and varied among locations with 12%, 29%, and 34% at BE, RR, and DC, respectively (Table 9). The highest proportion of zero catch was in 2006 and accounted for 44%, 75%, and 51% of net sets at BE, DC, and RR, respectively.

In terms of AIC, the zero-augmented Hurdle model was selected as the best model and used as input to the SPM and iSCAM models, having the lowest Akaike information criterion (AIC) (20,785), although the ZINB model was similar (20,792) (Table 10). $AIC-AIC_{min}$ for Hurdle compared to ZINB was < 10 and AIC weight ($W_i$) for Hurdle and ZINB, were 96% and 4%, respectively. Among the three ZAMs (Hurdle, ZIP, and ZINB), both non-zero and zero-value catches were fit separately and the medians of Pearson residuals (-0.32 for Hurdle; -0.46 for ZIP; and -0.32 for ZINB) were greater than those in GLMs, particularly for Poisson (-0.76).

The estimated CPUEs from the Poisson model appeared similar to those from the Quasi-Poisson, Hurdle, ZIP, and ZINB models, with two peaks in 1998 and 2008 (Figure 12). However, the dispersion test for Poisson regression indicated the existence of an over-dispersion (dispersion index = 8.03, $Z = 11.23$, $p < 0.0001$), which violates the underlying assumption of Poisson distribution with equal mean to its variance. During 1996–2014, the average standardized CPUE (mean ± SE) based on AIC weights (96% Hurdle and 4% ZINB) was 7.75 ± 0.18 fish per set (1.49 ± 0.19 ~ 23.86 ± 1.11 fish per set).

### Size and age composition

Through the implementation of exploratory and monitoring programs during 1986–2014, we found significant sexual differences in the fork length ($F = 4.67$, $p < 0.05$) and round weight ($F = 74.84$, $p < 0.0001$). The average (median ± standard deviation) fork length of males (450 ± 82 mm, range = 100–767 mm, $n = 3{,}230$) was roughly similar to the females (460 ± 64 mm, range = 84–667 mm, $n = 7{,}040$), while the average round weight of males (1,100 ± 699 g, range = 9–6,000 g) was 8% greater than females (1,060 ± 481 g, range = 6–4,450 g) (Table 11). Two-way ANOVA revealed that significant spatiotemporal variations of the fork length of Dolly Varden, collected during 1995–2014, were elucidated by years ($F = 92.21$, $p < 0.0001$, $df = 19$), sites ($F = 101.42$, $p < 0.0001$, $df = 2$) and strong year-site interactions ($F = 4.41$, $p < 0.0001$, $df = 38$). The annual medians of fork length were below the overall median prior to 1999, then fluctuated around the average for the remainder of the time series at RR, but at DC and BE length was above average from 2010–2013 (Figure 13). Fish captured at DC were slightly larger (average = 444 ± 83 mm, $n = 1{,}920$) compared to BE (average = 459 ± 66 mm, $n = 3{,}409$) and RR (average = 470 ± 76 mm, $n = 1{,}731$). Over the duration of the monitoring period, the grand average fork length was accounted for 449 ± 79 mm. Two-way ANOVA also indicated considerable spatiotemporal variations in the round weight by years ($F$ = 73.01, $p$ < 0.0001, $df$ = 19) and year-site interactions ($F$ = 17.41, $p$ < 0.0001, $df$ = 38). However, the difference in the round weight of the species is not statistically remarkable among the monitoring sites ($F$ = 2.22, $p$ = 0.11). Over 1995–2014, the grand median round weight was 1,089 ± 996 g.

Temporal variation in the age composition of Rat River Dolly Varden collected in 1986–2014 was depicted in Figure 14. The dominant classes (> 20%) during 1990–1991, 1996–2005, 2006–2012, and 2013–2014 were age classes 4–5, 5–6, 4–5, and 5–6, respectively. Ages 5 and 6 dominated the harvest prior to 2005 and in 2013 and 2014, in the intervening years ages 4–5 dominated. The total proportion of fish younger than 3 yrs or age 8 and older was ~ 8%. One-way ANOVA found the significant sexual differences in age composition ($F$ = 32.35, $p$ < 0.0001). Males ranged in age from 3–11 yrs with the average 5 ± 1 yrs ($n$ = 2,535), while females ranged from 3–13 yrs with the average 6 ± 1 yrs ($n$ = 4,499).

## Length-weight relationships and size-at-age growth

The sex-specific length-weight relationships for 13,549 Dolly Varden, collected for exploratory and monitoring programs from 1986 through 2014, were derived to:

male: $W = 3{,}15 \times 10^{-5} L^{2{,}8426}$ ($r^2$ = 0.89, $p$ < 0.0001, $n$ = 3,230)

female: $W = 1{,}28 \times 10^{-4} L^{2{,}6032}$ ($r^2$ = 0.79, $p$ < 0.0001, $n$ = 7,040)

combined: $W = 3{,}01 \times 10^{-5} L^{2{,}8377}$ ($r^2$ = 0.90, $p$< 0.0001, $n$ = 13,549)

Model parameters of length-weight relationships and growth in the fork length (mm) and round weight (g) were summarized in Table 11 and Figure 15. Because of under-represented samples for males < 3 and > 9 yrs of age and females < 3 and > 12 yrs (Figure 14), the size-at-age growth parameters were estimated for males between ages 3 and 9 and females between ages 3 and 11 (Table 11 and Figure 16). Males ≥ 4 years of age were larger than females for both length (6%) and round weight (17%).

## Natural mortality

As Dolly Varden grows, the age-dependent $M$ decreases in the early life stages and becomes relatively stable after age 7 (Table 12, Figure 17). Typically, $M$ values for age=0 and age=1 are higher than older ages due to greater vulnerability to predation, disease, or inclement environmental conditions. Among three empirical models, the length-at-age model (0.34 ± 0.10 per year), and the life history model (0.33 ± 0.04 per year) had similarly high $M$ (mean ± SE), followed by the weight-at-age model (0.22 ± 0.03 per year) (Table 12). The geometric mean for the three empirical models combined revealed that the $M$ values were 0.26 ± 0.05, 0.31 ± 0.05, and 0.29 ± 0.05 for males, females, and sexes combined, respectively.

# STOCK ASSESSMENT MODEL OUTPUTS

## Depletion-based stock reduction analysis

The average harvest (median ± SD: 1,493 ± 1,012 fish) during 1991–2005 was used for a prior catch input into DB-SRA, as was a period when the harvest was seen to be highly variable, with near the historical lows at the beginning and end, and a series high in 1998. Model output is given in Table 13 and includes the estimates of DCAC (1,144 ± 183 fish), $K$ (18,804 ± 5,790 fish), and relative stock status (delta = 0.60 ± 0.10).

The following population dynamics parameters (median ± SD) were derived: maximum sustainable yield (MSY) (1,217 ± 312 fish), number of fish > 440 mm at MSY ($N_{MSY}$) (7,466 ± 2,357 fish), fishing mortality at MSY ($F_{MSY}$) (0.21 ± 0.11 per year) and exploitation rate at MSY ($E_{MSY}$) (0.17 ± 0.06 per year) (Table 13). The average harvest during 1991–2005 was (36% above MSY and the mean DCAC was slightly below the MSY level (Figure 18). The probability distribution of all model parameters was typically normal or log-normal (Figures 19 and 20). The Spearman correlations were statistically significant for three pairs of covariates: -0.89 for OFL and Delta, -0.72 between $K$ and $M$ (Figure 19), as well as 0.98 between $F_{MSY}$ and $E_{MSY}$ (Figure 20).

Fishable population abundance (fish > 440 mm) estimated by the model decreased from 18,804 ± 5,790 fish in 1986 to 6,732 ± 4,319 fish in 1999 and then steadily increased to 14,483 ± 4,891 fish predicted for 2015 (Figure 21). The overfishing limit behaved similarly, varying from 3,046 ± 773 fish in 1986 to 2,070 ± 886 fish predicted for 2015. The abundance status ($N/N_{MSY}$) changed from 2.52 in 1986 to 1.64 in 2014, with a low of 0.90 in 1999 (Figure 22). The fishing status ($F/F_{MSY}$) had the reverse trend, going from 0.25 in 1986 to 1.93 in 1998 and then dropping to 0.1360 in 2014. These results indicate that Rat River Dolly Varden was overfished from 1997 to 1999 and that the reduction in harvest since 2000 may have contributed to an improvement in population status.

## Surplus production model

The model-estimated CPUE was compared with the standardized CPUE (Hurdle model), showing two significant biased periods 1998–1999 and 2011–2014 (Figure 23). The former corresponded to the period of high catch but lowering the population abundance. The latter reflected the fact that the population stayed at the state of slow recovery as the voluntary reduction of the subsistence harvest. Using maximum likelihood estimation (MLE), two key time-invariant model parameters $K$ and $r$ (mean ± SD) were estimated as 22,093 ± 307 fish and 0.18 ± 0.03 per year, respectively. The key population dynamics parameters $N_{MSY}$, MSY, $F_{MSY,}$ and $E_{MSY}$ were estimated to 11,047 fish, 1,005 fish, 0.09 per year, and 0.24 per year, respectively (Table 14). Over twofold uncertainty was associated with observation error ($\tau = 0.29$) compared to process error ($\sigma = 0.12$).

Based on MCMC simulations, median values of $N_{MSY}$, MSY, $F_{MSY,}$ and $E_{MSY}$ were 13,689 ± 7,042 fish, 2,242 ± 1,385 fish, 0.16 ± 0.03 per year, and 0.13 ± 0.02 per year, respectively (Table 14). These median values were less than the mean values, indicating the distributions were positively skewed (Figure 24, Table 14). The Spearman correlations between covariates for the probability distributions of model parameters indicated a strong negative relationship between $K$ and the catchability coefficient ($q$) ($\rho = -0.66$, $p < 0.001$) and a strong positive correlation between MSY and $N_{MSY}$ ($\rho = 0.94$, $p < 0.001$) (Figure 24).

Temporal trends in the population abundance ($N$), $N/N_{MSY}$, and $F/F_{MSY}$ statuses indicated that the population experienced over-exploitation between 1996–1999 when population abundance decreased and $F/F_{MSY}$ was > 1.00 (Figure 25). The SPM indicated that since 2000 the population status has been in a healthy state because of strong productivity ($B/B_{MSY} > 2.00$) and very low fishing pressure ($F/F_{MSY} < 0.10$).

## Integrated statistical catch-at-age model

The source of the observation error was primarily (99%) CPUE ($\tau^2_{CPUE} = 0.44$) (39%) and age composition ($\tau^2_{Age} = 0.67$) (61%). Model process error ($\sigma^2 = 11.21\times10^{-4}$) was lower, mainly due to well-selected probability distribution and penalty functions. Log-transformed residuals of observed and predicted values were examined for three model inputs: 1) standardized CPUE

(Hurdle model), 2) subsistence harvest (Figure 26), and 3) age composition (Figure 27). The observed CPUEs were 6.36 ± 1.16 fish per set (1.29 ~ 22.52 fish per set) while the predicted CPUEs were 5.94 ± 0.84 fish per set (1.55 ~ 13.59 fish per set). The general trend of the estimated CPUEs was approximately 7% lower than the observed CPUEs (Figure 26a), except for two considerably biased CPUE observations in 1998 and 2008. There was no difference between observed and predicted estimates for harvest (Figure 26b) but there were remarkable differences in the catch-at-age proportions (Figure 27). Compared with those predicted age compositions, the significant observed catch-at-age proportions have been identified in age 6 in 1997 and ages 3 and 4 in 2011 and 2012. Considerable under-represented observations in catch-at-age proportions also occurred in ages 3–5 in 1999 and age 4 in 2013 and 2014. Since 2010, the age 6 cohort has dominated the estimated catch-at-age proportion (> 30%) of the Rat River Dolly Varden fishery (Figure 27).

Based on MCMC simulations, average (median ± SD) $N_{MSY}$, $B_{MSY}$, $F_{MSY}$, and $E_{MSY}$ were 23,965 ± 13,003 fish, 6,520 ± 1,293 kg, 0.25 ± 0.12 per year, and 0.20 ± 0.08 per year, respectively. MSY was estimated as 1,007 ± 715 fish and 1,660 ± 200 kg (Table 15). Given the estimate of time-invariant natural mortality, we found that both fishing mortality and exploitation rate for Rat River Dolly Varden had relatively higher values as $F > 0.10$ and $E > 0.08$ in 1996–1998 and 2003–2005 (Figure 28). Since 2006, the exploitation rate dropped to < 0.5. Associated with catch-at-age proportions, the estimated annual abundance (all age classes combined) varied between 513,376 (2005) and 853,760 (1998), with an average of 586,444 ± 91,667 individuals. Total biomass varied between 67,639 kg (2003) and 302,567 kg (1998), with an average of 103,209 ± 60,985 kg. Including 50% age 6 and all fish age 7 and older, the SSB varied between 1,370 kg (2006) and 13,396 kg (2001) with an average of 5,004 ± 3,914 kg, while the recruits changed from 335,752 individuals (2007) to 384,210 individuals (2000) with an average of 379,477 ± 15,699 individuals. Over the monitoring period, two different response patterns between SSB and recruits can be found. In 1998–2001, SSB was double the average but annual recruits were just slightly above the average (Figure 29). The greater SSB did not contribute to a stronger cohort of recruits because a large amount of mature (ages 6+) fish were harvested as a subsistence food source (Figures 26 and 27). Since the implementation of voluntary closure or reduction of subsistence fisheries in 2006, SSB gradually increased to 8,219 kg in 2011 and recruits rebounded to as high as 622,184 individuals in 2010.

Stock status indicator $B/B_{MSY}$ decreased from 1996 to below the reference threshold $B/B_{MSY} = 1$ in 2004 (i.e., over-exploited state), increasing again to just above the reference line in 2011 before declining again through to 2014. (Figure 30). High $F/F_{MSY}$ occurred between 1996 and 2005, indicating a state of overfishing. $SSB/SSB_{MSY}$ has been in an overfished state except for 1996 and 1997 (Figure 30).

The probability distributions of most population dynamics model parameters followed somewhat normal or log-normal distributions (Figures 31 and 32). The exceptions were: 1) right-skewed or log-normal distribution for the stock-recruitment parameter ($h$); 2) left-skewed or normal distribution for $B_{MSY}$ and $F_{MSY}$; 3) beta distribution for the initial value of recruits ($r_{init}$); and, 4) bimodal distribution for fraction of the total variance ($\rho$). Spearman correlations of most covariates were rather weak, indicating typical non-linearity. However, there were significant positive correlations between $B_0$ and SSB ($r = 0.83$, $p < 0.001$), MSY and $F_{MSY}$ ($r = 0.81$, $p < 0.001$), and $r_0$ and $\bar{r}$ ($r = 0.62$, $p < 0.05$), which supports the existence of a spawner-recruitment relationship. The significant negative correlation between $B_{MSY}$ and $F_{MSY}$ ($r = -0.87$, $p < 0.001$) indicates that total biomass is greater at lower fishing intensity.

## Multimodel averaging

Of the three quantitative fisheries assessment models, SPM produced the highest $N_{MSY}$ and MSY and the lowest $F_{MSY}$ and $E_{MSY}$ estimates. The SCA model produced the lowest $N_{MSY}$, a moderate abundance MSY, and the highest $F_{MSY}$ and $E_{MSY}$. DB-SRA generated intermediate $N_{MSY}$, $F_{MSY}$, and $E_{MSY}$ estimates and the lowest MSY (Table 15). Inverse variance weighting was used to estimate parameter means among the three models. Median values for abundance parameters (± SD) were: $N_{MSY}$ (10,813 ± 1555 fish), MSY (1,301 ± 188 fish), and $F_{MSY}$ (0.18 ± 0.02 per year), and $E_{MSY}$ (0.15 ± 0.01 per year) (Table 15). Median values for biomass parameters were: $B_{MSY}$ (6,520 ± 1,293 kg), MSY (1,660 ± 200 kg), $F_{MSY}$ (0.25 ± 0.12 per year), and $E_{MSY}$ (0.20 ± 0.08 per year) and spawning stock biomass at MSY ($SSB_{MSY}$) was 643 ± 310 kg per year (Table 15).

# DISCUSSION

## DATA EXPLORATION OF ENVIRONMENTAL VARIABLES USED IN THE STANDARDIZATION OF CPUE

Contemporary data exploration and interpretation have advanced our understanding of data quality issues, including some common issues of outliers, collinearity, and zero-inflated attributes in natural resource research (McCullagh and Nelder 1989, Martin et al. 2005, Zuur et al. 2010). In our study, we found outliers accounted for ~ 1% of the data and mainly resulted from 1) variable environmental conditions; 2) inconsistent sampling schedule; 3) variable gill net panel length and mesh size; 4) change of fishing duration; and, 5) incorrectly recorded data. The negative correlation between DOY and water temperature and the positive correlation between water level and turbidity indicated the possible confounding effects of these variables on the seasonal migration behaviour for Rat River Dolly Varden. We chose to remove variables DOY and water level and retain temperature and water turbidity in the set of explanatory variables for CPUE standardization to minimize the extent of collinearity in the regression coefficients (McCullagh and Nelder 1989, Belsley 1991, Dobson, 2002). Our examination of these results concerning collinearity concur with other observations; that the upstream migration of this species is strongly influenced by local temperature (Bond and Quinn 2013) and habitat attributes like water turbidity (Mochnacz et al. 2010, 2020, Fenkes et al. 2016). Thus, we recommend water temperature and Secchi disc depth (to measure water turbidity) be among the environmental variables included in any catch monitoring program for Rat River Dolly Varden.

The occurrence of as high as 26% zero catch in the Rat River Dolly Varden CPUEs may be related to habitat suitability (Bond and Quin 2013, Mochnacz et al. 2020), or cohort strength and biological fitness, as well as changes in fishing protocols for the community-based monitoring operation. The CPUE for all three locations combined was highest in 1998 and 2008 but the factors driving these peaks is not known. In 1998 migration began earlier than in most years, July 24 (DOY 205), and extended to September 7 (DOY 251) due to it being an El Niño year. However, the 2008 peak was driven by an increase in CPUE at the RR site (Figure 12), where migration started late, from August 9 (DOY 222) to September 11 (DOY 255). We also found that 58% of Dolly Varden caught in 2008 were 5 yrs old indicating a strong 2003 year class. However, > 50% of fish caught in 1997 were 6 yrs old and CPUE was near average. Temperature had a significant negative effect on catch success (or zero catch) and CPUE. Warm water is associated with earlier downstream migration in the spring and longer feeding periods in coastal waters (Mochnacz et al. 2010, Bond and Quinn 2013, Morrison et al. 2021). The longer fish spend feeding along the Beaufort Sea coast the better their somatic condition and fitness (Bond and Quinn 2013, Kovach et al. 2013 Gallagher et al. 2013, 2020).

## DETECTION PROBABILITY AND MODEL SELECTION

GLMs cannot be directly applied to model observations that contain many zeroes because the natural logarithm of zero is undefined (i.e., infinity) (Zar 2010) and results could be biased, especially when count data is over-dispersed or has an excess number of zeros (McCullagh and Nelder 1989, Martin et al. 2005, Lindén and Mäntyniemi 2011). Because 27% of the catches in the harvest monitoring dataset were zero, the distribution of CPUE was skewed and the underlying assumption about Gaussian probability distribution was violated. However, an exponential family of distribution functions can be used to structure GLMs, such as Poisson regression, quasi-Poisson regression, or negative binomial regression (Battaile and Quinn 2004, Bishop et al. 2004, Brodziak and Walsh 2013).

When applying multiple candidate GLMs to the CPUE standardization, we found that a negative binomial distribution was more appropriate than either Poisson or quasi-Poisson distribution models. Furthermore, among ZAMs, both Hurdle and ZINB models, structured with negative binomial probability distributions, were superior to ZIP for quantifying variation in catch, which is similar to those published studies by Lewin et al. (2010), Lynch et al. (2012), and Brodziak and Walsh (2013). Between Hurdle and ZINB, the Hurdle model has properties that best suited our data, suggesting that it was the preferred model for standardizing our CPUE data.

## STOCK STATUS AND SUSTAINABILITY OF FISHERIES

We explored DB-SRA, SPM, and SCA models to assess the population dynamics and delineate temporal changes of abundance, SSB, stock and fishing status for Rat River Dolly Varden. Applying a multimodel averaging approach (WIV), we estimated for the first time population dynamics parameters: MSY (1,472 ± 188 fish), $N_{MSY}$ (6,422 ± 634 fish) and $F_{MSY}$ (0.18 ± 0.02 per year) for Rat River Dolly Varden.

Short periods of overfishing were indicated by the DB-SRA model during 1998–1999 ($F/F_{MSY} > 1$) and by the SPM during 1996 to 1999. However, the SCA model indicated the stock was overfished ($B < B_{MSY}$) from 2002 to 2007, and then again from 2012 to 2014. Given 2 out of 3 models indicates overfishing in the late 1990s we conclude that the Rat River Dolly Varden stock was likely overharvested during this period. Since the voluntary closure or reduction in harvest in 2006, the population has shown signs of a slow increase in abundance.

For fisheries stock assessment, it is critical to select models that are suitable and fit the available data. Of three candidate models, DB-SRA requires a time series of harvest and information on natural mortality and age at maturity. It is considered fairly robust for data-poor fisheries, but sensitive to the time period selected and the assumed distribution of the starting biomass (Dick and MacCall 2011, Wetzel and Punt 2011). SPM is a biomass dynamics model that is commonly used to fit CPUE time series and harvest information, assuming fishing mortality is less than natural mortality (Hilborn and Walters 1992). Age-based models like SCA are more complex quantitative models that can disseminate multiple sets of fisheries monitoring information to derive estimates of historical population size and fishing mortality. SCA assumes that removals (catch) are known without error and many SCA models also assume constant catchability and age-specific natural mortality (Quinn and Deriso and Quinn 1999). The analysis presented here is the first attempt at analyzing subsistence catch data for Rat River Dolly Varden. While there are uncertainties with the estimates from these models they all indicate that harvest is below MSY. Resource managers can now use this information to inform decisions concerning the Rat River Dolly Vardon stock.

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# TABLES AND FIGURES

*Table 1. Description of variables used to standardize CPUE, measured during the Rat River Dolly Varden monitoring program (1996–2014).*

| Variable | Description | Variable Type |
|---|---|---|
| **Response variable** | | |
| Catch number | Number between 0 and 56 Dolly Varden caught using a monitoring gillnet set | Discrete integer |
| **Explanatory variables** | | |
| Year | Year 1996–2014 | Categorical |
| DOY | Day of the year was generated in terms of sampling day, month, and year, and grouped by every ten days between DOY 210 and 270. | Discrete integer |
| Location | Sampling locations 1: Big Eddy (BE), 2: Destruction City (DC), and 3: mouth of Rat River (RR) | Categorical |
| Water level | 1: low, 2: normal, and 3: high | Categorical |
| Water colour | clear, cleaning, and dirty | Categorical |
| Water debris | None, fewer or more | Categorical |
| Turbidity | Combination of water colour and content of debris: 1: clear without debris, 2: clear with fewer debris, 3: clear with lot debris, 4: dirty without debris, 5: dirty with fewer debris and 6: dirty with more debris | Categorical |
| Mesh size | 89 mm (3.5"), 102 mm (4"), 114 mm (4.5"), 127 mm (5") and 140 mm (5.5") | Categorical |
| Net number | Number of nets used | Continuous |
| Net length | Net length measured by the yard and converted to meter, and grouped by every 5 m between 1 to 30 m. | Continuous |
| Soak time | Hours of fishing when the net stayed in the water, grouped as < 5, 5.1–10, 10.1–15, 15.1–20, 20.1–25 | Continuous |
| Temperature | Water temperature at the time of monitoring set | Continuous |
| Fork length | The length of a fish measured from the tip of the snout to the end of the middle caudal fin rays in mm | Continuous |
| Round weight | The weight of the whole fish before processing or removal of any part. | Continuous |
| Sex | Male, female or un-identified | Categorical |
| Age | Reading otoliths | Discrete integer |

*Table 2. Expression of mathematical models used for catch-per-unit effort standardization: three generalized linear models (Posisson, Quasi-Poisson and negative binomial regression); three zero-augment models, (Hurdle, zero-inflated Poisson, and zero-inflated negative binomial), as well as the multimodel inference.The model parameters p, μ, θ represent probability, mean CPUE, and shape parameters, respectively. Model coefficients $\beta_0$ and $\gamma_0$ are random intercepts, and $\boldsymbol{\beta}$ and $\boldsymbol{\gamma}$ are vectors of random effects. Variables **x** are covariate vectors of interest. Multimodel inference is based on the difference (Δ) of corrected Akaike information criterion (AIC ) ($AIC_c$) and minimum AIC ($AIC_{min}$), AIC weight ($w_i$) and $\beta_{ij}$. Here subscript ij are ith model and j year, respectively.*

| Parameter | Probability distribution function |
|---|---|
| **Generalized linear model (GLM)** | |
| Poisson | $P(x, y_i = \mu) = \frac{\mu^x e^{-\mu}}{x!}$ |
| Negative binomial or Quasi-Poisson | $P(y; \mu, \theta) = \frac{\Gamma(y+\theta)}{\Gamma(\theta) \times y!} \times \frac{\mu^y \theta^\theta}{(\mu+\theta)^{y+\theta}}$ |
| | $u_i = exp(\beta_0 + \beta_1 x_{1,x} + \ldots \beta_p x_{p,i})$ |
| **Zero-augment model (ZAM)** | |
| Hurdle | $P(y_i = 0) = 1 - p_i$<br>$P(y_i = k) = p_i \frac{\mu^k e^{-\mu}}{k!(1-e^{-\mu})}$ |
| Zero-inflated Poisson (ZIP) | $P(y_i = 0) = (1 - p_i) + p_i e^{-\mu}$<br>$P(y_i = k) = p_i \frac{\mu^k e^{-\mu}}{k!}$ |
| Zero-inflated negative binomial (ZINB) | $P(y_i = 0) = p_i + (1 - p_i) \frac{\theta^\theta}{(\mu_i + \theta)^\theta}$<br>$P(y_i = k) = (1 - p_i) \frac{\Gamma(k+\theta)}{\Gamma(\theta) k!} \times \frac{\mu_i^k \theta^\theta}{(\mu_i + \theta)^{k+\theta}}$ |
| | $\text{logit}(p_i) = \ln\left(\frac{p_i}{1-p_i}\right) = \beta_0 + \beta_1 x_{1i} +,,, + \beta_n x_{ni}$<br>$log(\mu_i) = \gamma_0 + \gamma_1 x_{1i} +,,, + \gamma_n x_{ni}$ |
| Multimodel inference (MMI) | $\Delta_i = AICc - AICc_{min}$<br>$w_i = \frac{e^{-\frac{1}{2}\Delta_i}}{\sum_{i=1} e^{-\frac{1}{2}\Delta_i}}$<br>$\beta_{ij}^{'} = \beta_{ij} \times w_i$ |

*Table 3. Number of Dolly Varden sampled at Big Eddy (BE), Destruction City (DC), the mouth of Rat River (RR), and Fish Creek (FC), Northwest Territories, by gear type (electro = electroshock) for biological observation, 1986–2014.*

| Year | BE | DC | RR | FC | | | RR | Sum |
|---|---|---|---|---|---|---|---|---|
| | | Gillnet | | Angling | Electro | Seine | Hoopnet | |
| 1986 | - | - | 110 | - | 57 | - | - | 167 |
| 1988 | - | - | | - | 119 | - | - | 119 |
| 1989 | - | - | 508 | - | - | - | 1084 | 1,592 |
| 1990 | - | - | 165 | - | - | - | - | 165 |
| 1991 | - | - | 61 | - | - | - | - | 61 |
| 1992 | - | - | 220 | - | - | - | - | 220 |
| 1993 | - | - | 329 | - | - | - | - | 329 |
| 1994 | - | - | 196 | - | - | - | - | 196 |
| 1995 | 557 | 178 | 122 | - | - | 444 | - | 1,301 |
| 1996 | 392 | 198 | 142 | - | - | - | - | 732 |
| 1997 | 441 | 100 | 141 | - | - | 445 | - | 1,127 |
| 1998 | 387 | 180 | 100 | - | - | - | - | 667 |
| 1999 | 237 | 100 | 99 | - | - | - | - | 436 |
| 2000 | 252 | 139 | 110 | - | - | - | - | 501 |
| 2001 | 245 | 153 | 118 | 1 | - | 414 | - | 931 |
| 2002 | 131 | 111 | 125 | - | - | - | - | 367 |
| 2003 | 199 | 92 | 100 | - | - | - | - | 391 |
| 2004 | 53 | 85 | 88 | 2 | - | 280 | - | 508 |
| 2005 | 58 | 99 | 98 | - | - | - | - | 255 |
| 2006 | 40 | 39 | 42 | - | - | - | - | 121 |
| 2007 | 40 | 40 | 40 | - | - | 432 | - | 552 |
| 2008 | 40 | 39 | 40 | - | - | 524 | - | 643 |
| 2009 | 46 | 75 | 71 | - | - | 470 | - | 662 |
| 2010 | 74 | 72 | 75 | - | - | 471 | - | 692 |
| 2011 | 47 | 50 | 50 | - | - | 157 | - | 304 |
| 2012 | 50 | 50 | 50 | - | - | - | - | 150 |
| 2013 | 60 | 60 | 60 | - | - | - | - | 180 |
| 2014 | 60 | 60 | 60 | - | - | - | - | 180 |
| Total | 3,649 | 1,920 | 4,126 | 68 | 177 | 3,696 | 1,084 | 13,549 |

*Table 4. Notation for the depletion-based stock reduction model equations.*

| Model | Equation | Reference |
|---|---|---|
| Depletion-corrected average catch | $DCAC = \frac{\sum C_t}{n + \frac{\Delta}{\left(\frac{B_{MSY}}{B_0}\right)\left(\frac{F_{MSY}}{M}\right)M}}$ | MacCall 2009 |
| | $Bt = Bt - 1 + P(Bt - a) - Ct - 1$<br>$P(B_{t-a}) = gMSY\left(\frac{B_{t-a}}{K}\right) - gMSY\left(\frac{B_{t-a}}{K}\right)^n$<br>Here, $g = \frac{n^{n/(n-1)}}{n-1}$ (n > 0) | Pella and Tomlinson 1969, Fletcher 1978, McAllister et al. 2000, Walters et al. 2006 |
| | $P = B_{t-a}\left(P(B_{join})/B_{t-a} + s(B_{t-a} - B_{join})\right)$<br>$s = (1-n)gmB_{join}^{n-2}K^{-n}$ | Hilborn and Walters 1992 |
| Stock reduction analysis | $E = \frac{F}{M+F}\left(1 - e^{-(F+M)}\right)$ | |

| Symbol | Description |
|---|---|
| $t$ | Time step year $t$ = 1,2,…,$T$ |
| Observations | |
| $C_t$ | Harvest landing in year $t$ |
| Model parameters | |
| *DCAC* | Depletion-corrected average catch |
| $\Delta$ | Relative stock status |
| $B_t$, $B_{join}$, | Biomass by year t or joint point |
| $B_0$, $B_{t-a}$ | Biomass at virgin or at entry to the reproduction |
| $K$ | Virgin population size for growth or biological carrying capacity |
| $F$ | Fishing mortality rate |
| $M$ | Natural mortality rate |
| $E$ | Exploitation rate |
| $n$ | Shape parameter $n$ = *number of years in catch time series* |
| Derived parameters | |
| MSY | Maximum sustainable yield |
| $B_{MSY}$ | Biomass at MSY |
| $F_{MSY}$ | Fishing mortality at which MSY is obtained |
| $P$ | Latent annual Production |
| $s$ | slope of the PTF production-to-biomass ratio |
| $g$ | Shape index |

*Table 5. Notation for the surplus production model.*

| Parameter | Equation |
|---|---|
| Model parameter | $\Theta = \{K, r, q, \sigma^2, \tau^2\}$ |
| Surplus production model | $\frac{dB_t}{dt} = rB_t\left(1 - \frac{B_t}{K}\right) - C_t$ <br> $Pt = Bt/K$ <br> $\hat{I}_t = q \times B_t$ <br> $P_{t+1} = P_t + rP_t(1 - P_t) - \frac{C_t}{K}$ |
| Derived management quantities | $F_{MSY} = r/2$ <br> $B_{MSY} = K/2$ <br> $MSY = (B_{MSY}) \times (F_{MSY})) = r \times K/4$ |

| Symbol | Description |
|---|---|
| Indices | |
| *t* | Time step in year *t* = 1,2,…,*T* |
| Observations | |
| $C_t$ | Harvest landing in year *t* |
| $I_t$ | Catch per unit effort (CPUE) in year *t* |
| Model parameters | |
| $B_t$ | Estimated exploitable biomass in year *t* |
| *K* | Virgin population size for growth or biological carrying capacity |
| *r* | Intrinsic population growth rate |
| $E_t$ | Fishing effort in year *t* |
| *q* | Gear-specific catchability coefficient |
| $P_t$ | Depletion rate parameter for $B_t/K$ |
| *σ* | Standard error in processing log-depletion rate |
| *τ* | Standard error in observed biomass indices |
| Derived parameters | |
| MSY | Maximum sustainable yield |
| $B_{MSY}$ | Biomass at MSY |
| $F_{MSY}$ | Fishing mortality at which MSY is obtained |

*Table 6. Notation for the integrated statistical catch-at-age model.*

| Parameter | Equation |
|---|---|
| Life-history schedule | $L_a = L_\infty\left(1 - e^{-K(a-t_0)}\right)$<br>$\mathrm{wa} = \mathrm{wla} \times (L_a)^{\mathrm{wlb}}$<br>$m_a = \frac{a^{u_1}}{a^{u_1} + u_2^{u_1}}$<br>$M_a = 1.69 w_a^{-0.292}$<br>$v_a = \frac{1}{1 + e^{-\frac{\hat{a}-a}{\gamma}}}$<br>$f_a = \frac{w_a}{1 + e^{-\frac{\square a - a}{\square \gamma}}}$ |
| Population dynamics | $N_1^a = \overline{R} e^{-\sum_{a=1}^{A-1} Z_{1,a} + \gamma_a}; \gamma \sim N\left(0, \sigma_\gamma^2\right)$<br>$N_{t+1,a+1} = N_{t,a} e^{-Z_{t,a}}$<br>$N_{t+1,A} = N_{t,A-1} e^{-Z_{t,A-1}} + N_{t,A} e^{-Z_{y,A}}$<br>$B_t = \sum_a N_{t,a} w_a$<br>$Z_{t,a} = M_a + F_{t,a} v_{t,a}$<br>$F_{t,a} = q_t E_t s_a$<br>$S_t = \sum_{a=1}^{A} m_a B_{t,a}$<br>$\widehat{C_{t,a}} = \left[\frac{F_{t,a}}{Z_{t,a}}(1 - e^{-Z_{t,a}}) N_{t,a}\right] \times w_a$<br>$\widehat{P_{t,a}} = \frac{\widehat{C_{t,a}}}{\widehat{C_t}}$ |
| Residuals and objective functions | $\mathrm{Catch}: \eta_t = ln(C_t) - ln\left(\hat{C}_t\right)$<br>$\mathrm{nll} = \sum\left[T\, ln(\sigma_c) + \frac{\sum_t(\eta_t^2)}{2\sigma_c^2}\right]$<br>$CPUE: v_t = \sum_a N_{t,a} e^{-\lambda_t Z_{t,a}} v_a w_a$<br>$\varepsilon_t = ln(I_t) - ln(q) + ln(v_t)$<br>$\mathrm{nll} = \sum_{t \in I_t} ln(I_t) + \frac{\varepsilon_t^2}{2\sigma_t^2}$<br>$\eta_{t,a} = ln(p_{t,a}) - ln(\widehat{p_{t,a}}) - \frac{1}{A}\sum_{a=1}^{A}\left[\left[ln(p_{t,a}) - ln(\widehat{p_{t,a}})\right]\right]$<br>$\hat{\tau}^2 = \frac{1}{(A-1)T}\sum_{t=1}^{T}\sum_{a=1}^{A}\eta_{t,a}^2$<br>$nll = (A-1)T\, ln(\hat{\tau}^2)$ |

*Table 7. Symbols and corresponding descriptions used in the notation of the integrated statistical catch-at-age model.*

| Symbol | Description |
|---|---|
| Indices | |
| $t$ | Time step year $t$=1,2,…,$T$ |
| $a$ | Age classes in years a={1,2,…,A} |
| Observed data | |
| $I_t$ | Survey abundance index in year $t$ |
| $C_t$ | Catch in kilogram in year $t$ |
| $p_{t,a}$ | The proportion of catch-at-age $a$ and year $t$ |
| $wla$ | Length-weight relation parameter |
| $wlb$ | Length-weight relation parameter |
| $L_\infty$ | Asymptotic length in mm |
| $K$ | Brody growth rate |
| $t_0$ | Theoretical age when length approaches zero |
| Derived parameter | |
| $l_a$ | Length-at-age $a$ |
| $w_a$ | Weight-at-age $a$ |
| $m_a$ | Maturity-at-age $a$ |
| $M_a$ | Natural mortality-at-age $a$ |
| $v_a$ | Age-specific vulnerability |
| $f_a$ | Mean fecundity-at-age |
| $r_t$ | Recruitment in year $t$ |
| $\zeta_a$ | Age-specific survivorship |
| $q$ | Survey catchability per fishing effort |
| $s_a$ | Selectivity-at-age $a$ |
| $\mu_1$ | Age-at-50% maturity |
| $\mu_2$ | Maturity-at-age function slope |
| $\tau_1$ | Coefficient of variation for survey abundance index |
| $\tau_2$ | Standard error in observed proportion-at-age |
| $\sigma$ | Standard error of log-transformed recruitment deviations |
| State variables | |
| $N_{t,a}$ | Abundance in age $a$ and year $t$ |
| $B_{t,a}$ | Biomass in age $a$ and year $t$ |
| $F_{t,a}$ | Fishing mortality in age $a$ and year $t$ |
| $Z_{t,a}$ | Total mortality in age $a$ and year $t$ |
| $N_t$ | Abundance in year $t$ |
| $B_t$ | Biomass in year $t$ |
| $S_t$ | Spawning biomass in year $t$ |

*Table 8. Subsistence harvest (number of fish) of Rat River Dolly Varden, used as input to population models.*

| Location | Shingle Point | Inuvialuit Aklavik | 50% harvest in Shingle Point assumed [b] | Gwich'in Delta | Unspecified | Rat River |
|---|---|---|---|---|---|---|
| 1986 | - | - | 50 | - | 1,050 | 1,100 |
| 1987 | 259 | - | 125 | - | 3,000 | 3,125 |
| 1988 | 147 | - | 74 | - | 1,607 | 1,681 |
| 1989 | 105 | 132 | 53 | 1,814 | - | 1,999 |
| 1990 | 214 | 60 | 107 | 884 | - | 1,051 |
| 1991 | 7 | 20 | 3 | 353 | - | 376 |
| 1992 | 17 | 250 | 9 | 775 | - | 1,034 |
| 1993 | 119 | 381 | 60 | 968 | - | 1,409 |
| 1994 | 33 | 842 | 17 | 1,296 | - | 2,155 |
| 1995 | 63 | 377 | 32 | 1,115 | - | 1,524 |
| 1996 | 805 | 547 | 403 | 1,970 | - | 2,920 |
| 1997 | 123 | 643 | 62 | 2,688 | - | 3,393 |
| 1998 | 576 | 255 | 386 | 3,119 | - | 3,760 |
| 1999 | 250 | 255 | 125 | 1,561 | - | 1,941 |
| 2000 | - | 8 | 0 | 1,485 | - | 1,493 |
| 2001 | - | - | - | 1,781 | - | 1,781 |
| 2002 | - | - | - | 635 | - | 635 |
| 2003 | - | - | - | 1,492 | - | 1,492 |
| 2004 | - | - | - | 772 | - | 772 |
| 2005 | - | - | - | 658 | - | 658 |
| 2006 | 127 | - | 64 | 124 | - | 188 |
| 2007 | 106 | 166 | 53 | 120 | - | 339 |
| 2008 | 29 | 56 | 15 | 119 | - | 190 |
| 2009 | 307 | 28 | - | 233 | - | 261 |
| 2010 | 240 | 13 | - | 266 | - | 279 |
| 2011 | 193 | 87 | 24 | 259 | - | 370 |
| 2012 | 412 | 45 | 21 | 300 | - | 366 |
| 2013 | 115 | 72 | 33 | 221 | - | 326 |
| 2014 | 404 | 56 | 40 | 291 | - | 387 |

a) The estimates by Harwood (2001).

b) 50% of the Dolly Varden caught at Shingle Point, that are considered from the Rat River population, as described by Harwood (2001).

c) Rat River Dolly Varden harvest estimated for 1972–2010 by Roux et al. (2012) and harvest in 2011–2014 calculated using mixed-stock fishery methods (Gallagher et al. 2015).

*Table 9. Total catch, zero-catch and % of sets with zero catch for Rat River Dolly Varden sampled from Big Eddy (BE), Destruction City (DC), and the mouth of Rat River (RR), Northwest Territories.*

| Year | BE | | | DC | | | RR | | | Overall | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Zero | Total | % | Zero | Total | % | Zero | Total | % | Zero | Total | % |
| 1996 | - | 58 | 0.00 | 6 | 106 | 5.66 | 40 | 95 | 42.11 | 46 | 259 | 17.76 |
| 1997 | - | 61 | 0.00 | 20 | 160 | 12.50 | 8 | 55 | 14.55 | 28 | 276 | 10.14 |
| 1998 | - | 57 | 0.00 | - | 67 | 0.00 | 1 | 26 | 3.85 | 1 | 150 | 0.67 |
| 1999 | 1 | 61 | 1.64 | 17 | 106 | 16.04 | 1 | 18 | 5.56 | 19 | 185 | 10.27 |
| 2000 | 13 | 60 | 21.67 | 47 | 112 | 41.96 | 33 | 97 | 34.02 | 93 | 269 | 34.57 |
| 2001 | 2 | 142 | 1.41 | 46 | 154 | 29.87 | 70 | 192 | 36.46 | 118 | 488 | 24.18 |
| 2002 | 8 | 63 | 12.70 | 26 | 88 | 29.55 | 30 | 88 | 34.09 | 64 | 239 | 26.78 |
| 2003 | 5 | 59 | 8.47 | 29 | 56 | 51.79 | 33 | 82 | 40.24 | 67 | 197 | 34.01 |
| 2004 | 15 | 67 | 22.39 | 52 | 90 | 57.78 | 38 | 87 | 43.68 | 105 | 244 | 43.03 |
| 2005 | 19 | 60 | 31.67 | 59 | 118 | 50.00 | 22 | 62 | 35.48 | 100 | 240 | 41.67 |
| 2006 | 18 | 41 | 43.90 | 69 | 92 | 75.00 | 23 | 45 | 51.11 | 110 | 178 | 61.80 |
| 2007 | 6 | 36 | 16.67 | 32 | 61 | 52.46 | - | 24 | 0.00 | 38 | 121 | 31.40 |
| 2008 | 1 | 27 | 3.70 | 9 | 30 | 30.00 | 2 | 31 | 6.45 | 12 | 88 | 13.64 |
| 2009 | 10 | 36 | 27.78 | 19 | 57 | 33.33 | 24 | 63 | 38.10 | 53 | 156 | 33.97 |
| 2010 | 3 | 42 | 7.14 | 41 | 87 | 47.13 | 4 | 49 | 8.16 | 48 | 178 | 26.97 |
| 2011 | - | 31 | 0.00 | 15 | 61 | 24.59 | - | 33 | 0.00 | 15 | 125 | 12.00 |
| 2012 | - | 35 | 0.00 | 20 | 60 | 33.33 | 5 | 38 | 13.16 | 25 | 133 | 18.80 |
| 2013 | 18 | 57 | 31.58 | 23 | 63 | 36.51 | 7 | 44 | 15.91 | 48 | 164 | 29.27 |
| 2014 | 9 | 48 | 18.75 | 22 | 70 | 31.43 | 5 | 54 | 9.26 | 36 | 172 | 20.93 |
| Total | 128 | 1041 | 12.30 | 552 | 1638 | 33.70 | 346 | 1183 | 29.25 | 1026 | 3862 | 26.57 |

*Table 10. Summary of estimate (Mean), standard error (SE), and significance (Sig) of coefficients of general linear models and zero-augmented models used to standardize Rat River Dolly CPUE. The significance levels are p < 0.0001 (***), p < 0.001 (**), p < 0.05 (*). Negative binomial (NB),the degrees of freedom (df), maximized log-likelihood (log L), Akaike information criterion (AIC), the difference between individual and minimum AIC weight ($w_i$), and observed and expected zero catches were included.*

| | Generalized linear model (GLM) | | | | | | | | | Zero-inflated model (ZAM) | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Poisson | | | Quasi-Poisson | | | NB | | | Hurdle | | | Zero-inflated Poisson | | | Zero-inflated NB | | |
| | Mean | SE | Sig | Mean | SE | Sig | Mean | SE | Sig | Mean | SE | Sig | Mean | SE | Sig | Mean | SE | Sig |
| **Positive catch coefficient** | | | | | | | | | | | | | | | | | | |
| intercept | -1.36 | 0.67 | * | -1.36 | 0.67 | * | -2.26 | 0.69 | *** | 0.72 | 0.62 | - | 0.68 | 0.66 | - | 0.78 | 0.66 | - |
| 1997 | 0.04 | 0.09 | - | 0.04 | 0.09 | - | 0.07 | 0.09 | - | 0.07 | 0.08 | - | -0.02 | 0.08 | - | 0.07 | 0.08 | - |
| 1998 | 0.87 | 0.08 | *** | 0.87 | 0.08 | *** | 0.91 | 0.08 | *** | 0.81 | 0.07 | *** | 0.72 | 0.07 | *** | 0.80 | 0.07 | *** |
| 1999 | 0.20 | 0.10 | * | 0.20 | 0.10 | * | 0.20 | 0.09 | * | 0.13 | 0.08 | - | 0.11 | 0.09 | - | 0.13 | 0.08 | - |
| 2000 | -0.58 | 0.10 | *** | -0.58 | 0.10 | *** | -0.65 | 0.10 | *** | -0.32 | 0.08 | *** | -0.29 | 0.08 | *** | -0.32 | 0.09 | *** |
| 2001 | -0.10 | 0.09 | - | -0.10 | 0.09 | - | -0.13 | 0.09 | - | 0.02 | 0.07 | - | -0.02 | 0.08 | - | -0.01 | 0.07 | - |
| 2002 | -0.67 | 0.11 | *** | -0.67 | 0.11 | *** | -0.64 | 0.10 | *** | 0.58 | 0.08 | *** | -0.57 | 0.10 | *** | -0.54 | 0.08 | *** |
| 2003 | -0.70 | 0.11 | *** | -0.70 | 0.11 | *** | -0.61 | 0.11 | *** | 0.35 | 0.10 | *** | -0.42 | 0.09 | *** | -0.33 | 0.10 | ** |
| 2004 | -1.08 | 0.11 | *** | -1.29 | 0.10 | *** | -1.33 | 0.10 | *** | 0.95 | 0.09 | *** | -0.90 | 0.08 | *** | -0.98 | 0.09 | *** |
| 2005 | -1.08 | 0.11 | *** | -1.08 | 0.11 | *** | -0.90 | 0.11 | *** | 0.70 | 0.10 | *** | -0.78 | 0.09 | *** | -0.69 | 0.10 | *** |
| 2006 | -1.84 | 0.14 | *** | -1.84 | 0.14 | *** | -1.85 | 0.14 | *** | -1.20 | 0.10 | *** | -1.09 | 0.10 | *** | -1.22 | 0.10 | *** |
| 2007 | -0.73 | 0.14 | *** | -0.73 | 0.14 | *** | -0.78 | 0.13 | *** | -0.59 | 0.12 | *** | -0.56 | 0.12 | *** | -0.57 | 0.11 | *** |
| 2008 | 0.69 | 0.15 | *** | 0.69 | 0.15 | *** | 1.05 | 0.15 | *** | 0.68 | 0.14 | *** | 0.55 | 0.13 | *** | 0.69 | 0.14 | *** |
| 2009 | -0.65 | 0.15 | *** | -0.65 | 0.15 | *** | -0.55 | 0.13 | *** | -0.57 | 0.10 | *** | -0.61 | 0.12 | *** | -0.60 | 0.11 | *** |
| 2010 | -0.57 | 0.12 | *** | -0.57 | 0.12 | *** | -0.51 | 0.11 | *** | -0.54 | 0.09 | *** | -0.59 | 0.10 | *** | -0.53 | 0.09 | *** |
| 2011 | 0.29 | 0.16 | - | 0.29 | 0.16 | - | 0.32 | 0.13 | * | 0.13 | 0.12 | - | 0.06 | 0.15 | - | 0.14 | 0.11 | - |
| 2012 | -0.19 | 0.13 | - | -0.19 | 0.13 | - | -0.08 | 0.12 | - | -0.14 | 0.10 | - | -0.24 | 0.11 | * | -0.14 | 0.10 | - |
| 2013 | -1.09 | 0.13 | *** | -1.09 | 0.13 | *** | -1.00 | 0.12 | *** | 0.96 | 0.11 | *** | -1.00 | 0.11 | *** | -0.94 | 0.10 | *** |
| 2014 | -0.35 | 0.19 | - | -0.35 | 0.19 | - | -0.16 | 0.16 | - | -0.34 | 0.14 | * | -0.39 | 0.18 | * | -0.36 | 0.13 | ** |
| Julian date | 0.01 | 0.00 | *** | 0.01 | 0.00 | *** | 0.02 | 0.00 | *** | 0.01 | 0.00 | *** | 0.01 | 0.00 | *** | 0.01 | 0.00 | *** |
| Net length | 0.00 | 0.01 | - | 0.00 | 0.01 | - | 0.01 | 0.01 | * | -0.02 | 0.01 | *** | -0.03 | 0.01 | ** | -0.02 | 0.01 | *** |
| Soak hour | -0.03 | 0.00 | *** | -0.03 | 0.00 | *** | -0.03 | 0.00 | *** | -0.04 | 0.00 | *** | -0.04 | 0.00 | *** | -0.04 | 0.00 | *** |
| Water temperature | 0.05 | 0.01 | *** | 0.05 | 0.01 | *** | 0.05 | 0.01 | *** | 0.03 | 0.01 | *** | 0.04 | 0.01 | *** | 0.03 | 0.01 | *** |
| Mesh 4.5" | -0.17 | 0.05 | ** | -0.17 | 0.05 | ** | -0.24 | 0.05 | *** | -0.18 | 0.04 | *** | -0.12 | 0.04 | ** | -0.18 | 0.04 | *** |
| Mesh 5.0" | -0.02 | 0.24 | - | -0.02 | 0.24 | - | -0.24 | 0.14 | - | 0.11 | 0.14 | - | -0.03 | 0.22 | - | -0.09 | 0.14 | - |
| No debris | 0.34 | 0.09 | *** | 0.34 | 0.09 | *** | 0.36 | 0.09 | *** | 0.26 | 0.08 | ** | 0.26 | 0.08 | ** | 0.26 | 0.08 | ** |
| Some debris | 0.13 | 0.09 | - | 0.13 | 0.09 | - | 0.14 | 0.09 | - | 0.12 | 0.08 | - | 0.12 | 0.08 | - | 0.12 | 0.08 | - |
| **Zero-inflated model coefficients (binomial with logit link)** | | | | | | | | | | | | | | | | | | |
| intercept | - | - | - | - | - | - | - | - | - | -8.46 | 1.31 | *** | 8.52 | 1.36 | *** | 9.97 | 1.76 | *** |
| 1997 | - | - | - | - | - | - | - | - | - | 0.34 | 0.28 | | -0.36 | 0.28 | | -0.33 | 0.34 | |
| 1998 | - | - | - | - | - | - | - | - | - | 3.03 | 1.03 | ** | -3.03 | 1.03 | ** | -4.09 | 3.90 | |
| 1999 | - | - | - | - | - | - | - | - | - | 0.51 | 0.30 | | -0.53 | 0.30 | | -0.48 | 0.35 | |
| 2000 | - | - | - | - | - | - | - | - | - | -1.28 | 0.23 | *** | 1.28 | 0.23 | *** | 1.42 | 0.28 | *** |

*Table 10 Continued*

| | Poisson | | | Quasi-Poisson | | | Negative Binomial | | | Hurdle | | | Zero-Inflated Poisson | | | Zero-Inflated NB | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Mean | SE | Sig | Mean | SE | Sig | Mean | SE | Sig | Mean | SE | Sig | Mean | SE | Sig | Mean | SE | Sig |
| 2001 | - | - | - | - | - | - | - | - | - | -0.38 | 0.22 | | 0.38 | 0.23 | | 0.55 | 0.28 | * |
| 2002 | - | - | - | - | - | - | - | - | - | -0.44 | 0.23 | | 0.43 | 0.23 | | 0.46 | 0.28 | |
| 2003 | - | - | - | - | - | - | - | - | - | -1.12 | 0.24 | *** | 1.12 | 0.24 | *** | 1.33 | 0.29 | *** |
| 2004 | - | - | - | - | - | - | - | - | - | -1.78 | 0.24 | *** | 1.70 | 0.24 | *** | 1.59 | 0.31 | *** |
| 2005 | - | - | - | - | - | - | - | - | - | -0.92 | 0.24 | *** | 0.89 | 0.24 | *** | 0.84 | 0.31 | ** |
| 2006 | - | - | - | - | - | - | - | - | - | -2.30 | 0.25 | *** | 2.22 | 0.25 | *** | 2.13 | 0.29 | *** |
| 2007 | - | - | - | - | - | - | - | - | - | -0.91 | 0.27 | *** | 0.88 | 0.27 | ** | 0.93 | 0.32 | ** |
| 2008 | - | - | - | - | - | - | - | - | - | 1.03 | 0.44 | * | -1.08 | 0.45 | * | -1.23 | 0.58 | * |
| 2009 | - | - | - | - | - | - | - | - | - | -0.30 | 0.28 | | 0.23 | 0.28 | | 0.02 | 0.36 | |
| 2010 | - | - | - | - | - | - | - | - | - | 0.03 | 0.28 | | -0.10 | 0.28 | | -0.25 | 0.34 | |
| 2011 | - | - | - | - | - | - | - | - | - | 1.39 | 0.33 | *** | -1.47 | 0.34 | *** | -1.65 | 0.38 | *** |
| 2012 | - | - | - | - | - | - | - | - | - | 0.23 | 0.32 | | -0.28 | 0.33 | | -0.39 | 0.41 | |
| 2013 | - | - | - | - | - | - | - | - | - | -0.63 | 0.28 | * | 0.49 | 0.29 | | 0.31 | 0.37 | |
| 2014 | - | - | - | - | - | - | - | - | - | 0.53 | 0.34 | | -0.62 | 0.38 | | -1.46 | 0.57 | * |
| Julian date | - | - | - | - | - | - | - | - | - | 0.02 | 0.00 | *** | -0.02 | 0.01 | *** | -0.02 | 0.01 | *** |
| Net length | - | - | - | - | - | - | - | - | - | 0.13 | 0.01 | *** | -0.14 | 0.01 | *** | -0.17 | 0.01 | *** |
| Soak hour | - | - | - | - | - | - | - | - | - | 0.05 | 0.01 | *** | -0.06 | 0.01 | *** | -0.09 | 0.01 | *** |
| Water temperature | - | - | - | - | - | - | - | - | - | 0.08 | 0.02 | *** | -0.08 | 0.02 | *** | -0.09 | 0.03 | *** |
| Mesh 4.5" | - | - | - | - | - | - | - | - | - | -0.44 | 0.11 | *** | 0.44 | 0.11 | *** | 0.51 | 0.15 | *** |
| Mesh 5.0" | - | - | - | - | - | - | - | - | - | -0.47 | 0.26 | | 0.45 | 0.29 | | 0.89 | 0.41 | * |
| No debris | - | - | - | - | - | - | - | - | - | 0.62 | 0.18 | *** | -0.60 | 0.19 | ** | -0.62 | 0.23 | ** |
| Some debris | - | - | - | - | - | - | - | - | - | 0.21 | 0.17 | - | -0.21 | 0.18 | - | -0.23 | 0.22 | - |
| df | | 27 | | | 27 | | | 28 | | | 55 | | | 54 | | | 55 | |
| logL | | -18576 | | | -18576 | | | -10822 | | | -10340 | | | -14230 | | | -10340 | |
| AIC | | 37205 | | | 37205 | | | 21701 | | | 20785 | | | 28573 | | | 20792 | |
| AIC-AINmin | | 16420 | | | NA | | | 916 | | | 0 | | | 7788 | | | 7 | |
| Wi | | 0 | | | NA | | | 0 | | | 0.96 | | | 0 | | | 0.04 | |
| Zero catch | | 1025 | | | 1025 | | | 1025 | | | 1025 | | | 1025 | | | 1025 | |
| Expected zero catch | | 119 | | | 119 | | | 764 | | | 1,025 | | | 1,025 | | | 1,032 | |

*Table 11. Summaries of statistical analysis of biological observation for Dolly Varden in the Rat River system, 1986–2014. a) basic statistics of fork length and round weight, including mean, median, standard deviation (SD), and range, as well as length-weight regression parameters like a and b values are regression coefficients, $r^2$ and n are squared correlation coefficient and sample size; and, b) von Bertalanffy growth model parameters including $L_\infty$, $W_\infty$, K, $t_0$, infinite fork length and round weight, Brody growth rate and age when fork length or round weight approach zero, σ is the standard deviation.*

*a)*

| | Male | Female | Combined |
|---|---|---|---|
| Fork length (mm) | - | - | - |
| Mean | 457 | 460 | 447 |
| Median | 450 | 460 | 449 |
| SD | 82 | 64 | 79 |
| Range | 100–767 | 84–667 | 57–767 |
| Round weight (g) | - | - | - |
| Mean | 1,262 | 1,160 | 1,089 |
| Median | 1,100 | 1,060 | 995 |
| SD | 699 | 481 | 561 |
| Range | 9–6,000 | 6–4,450 | 2–6,000 |
| Log(a) | -10.3648 | -8.9626 | -10.4119 |
| a | 3.15E-05 | 1.28E-04 | 3.01E-05 |
| b | 2.8426 | 2.6032 | 2.8377 |
| $r^2$ | 0.8892 | 0.7947 | 0.8998 |
| *n* | 3230 | 7040 | 13549 |

*b)*

| | Male | | Female | | All | |
|---|---|---|---|---|---|---|
| | Mean | SD | Mean | SD | Mean | SD |
| Fork length (mm)~Age (*t*: years) | | | | | | |
| $\log(L_\infty)$ | 6.5163 | 0.0524 | 6.3540 | 0.0355 | 6.4228 | 0.0538 |
| $\log(K)$ | -1.5509 | 0.0938 | -1.2069 | 0.0770 | -1.3921 | 0.1046 |
| $t_0$ | 0.0000 | 0.0001 | 0.0000 | 0.0001 | 0.0000 | 0.0001 |
| $\log(\sigma)$ | -2.9441 | 0.2357 | -3.0079 | 0.2357 | -2.7564 | 0.2357 |
| $L_\infty$ | 676.07 | - | 574.79 | - | 615.72 | - |
| $K$ | 0.2121 | - | 0.2991 | - | 0.2486 | - |
| $\sigma$ | 0.0526 | - | 0.0494 | - | 0.0635 | - |
| $n$ | 2,535 | - | 4,499 | - | 7,139 | - |
| Round weight (g)~Age (*t*: years) | | | | | | |
| $\log(W_\infty)$ | 8.3871 | 0.3091 | 7.8943 | 0.2636 | 8.0282 | 0.2971 |
| $\log(K)$ | -1.6916 | 0.1812 | -1.4609 | 0.1879 | -1.5034 | 0.1919 |
| $t_0$ | 0.0000 | 0.0001 | 0.0000 | 0.0001 | 0.0000 | 0.0001 |
| $\log(\sigma)$ | -1.2982 | 0.2357 | -1.2184 | 0.2357 | -1.1397 | 0.2357 |
| $W_\infty$ | 4390.07 | - | 2681.95 | - | 3,066.22 | - |
| $K$ | 0.1842 | - | 0.2320 | - | 0.2224 | - |
| $\sigma$ | 0.2730 | - | 0.2957 | - | 0.3199 | - |

*Table 12. Natural mortality for Rat River Dolly Varden, estimated for each age by sex (male and female) and for both sexes combined (all), using three empirical models (life-history, length-at-age, and weight-at-age). The geometric mean of these three models is also estimated.*

*a)*

| Age | Life history model (Chen and Watanabe 1989) | | | Length-at-age model (Gislason et al. 2010) | | |
|---|---|---|---|---|---|---|
| | Male | Female | All | Male | Female | All |
| 1 | 0.8011 | 0.9589 | 0.9307 | 1.5261 | 1.6986 | 1.7916 |
| 2 | 0.5233 | 0.6107 | 0.5802 | 0.7689 | 0.8215 | 0.8373 |
| 3 | 0.4087 | 0.4811 | 0.4485 | 0.5164 | 0.5596 | 0.5531 |
| 4 | 0.3472 | 0.4157 | 0.3811 | 0.3971 | 0.4423 | 0.4255 |
| 5 | 0.3095 | 0.3777 | 0.3410 | 0.3301 | 0.3790 | 0.3559 |
| 6 | 0.2845 | 0.3537 | 0.3152 | 0.2882 | 0.3410 | 0.3135 |
| 7 | 0.2671 | 0.3377 | 0.2977 | 0.2603 | 0.3166 | 0.2859 |
| 8 | 0.2545 | 0.3268 | 0.2835 | 0.2408 | 0.3003 | 0.2669 |
| 9 | 0.2448 | 0.3192 | 0.2758 | 0.2267 | 0.2891 | 0.2535 |
| 10 | 0.2380 | 0.3140 | 0.2696 | 0.2162 | 0.2812 | 0.2438 |
| 11 | 0.2326 | 0.3111 | 0.2647 | 0.2083 | 0.2755 | 0.2367 |
| 12 | 0.2284 | 0.3103 | 0.2610 | 0.2023 | 0.2715 | 0.2313 |
| 13 | 0.2254 | 0.3116 | 0.2584 | 0.1976 | 0.2685 | 0.2273 |
| 14 | 0.2233 | 0.3150 | 0.2568 | 0.1939 | 0.2664 | 0.2242 |
| 15 | 0.2223 | 0.3207 | 0.2563 | 0.1910 | 0.2648 | 0.2218 |
| 16 | 0.2223 | 0.3289 | 0.2567 | 0.1888 | 0.2637 | 0.2200 |
| Geomean | 0.2916 | 0.3779 | 0.3289 | 0.2991 | 0.3710 | 0.3351 |
| SE | 0.0385 | 0.0424 | 0.0443 | 0.0861 | 0.0917 | 0.1003 |

*b)*

| Age | Weight-at-age model (Lorenzen 1996) | | | Geometric mean | | |
|---|---|---|---|---|---|---|
| | Male | Female | All | Male | Female | All |
| 1 | 0.6418 | 0.5626 | 0.6140 | 0.9223 | 0.9713 | 1.0079 |
| 2 | 0.3882 | 0.3599 | 0.3778 | 0.5385 | 0.5652 | 0.5683 |
| 3 | 0.2974 | 0.2859 | 0.2938 | 0.3974 | 0.4254 | 0.4177 |
| 4 | 0.2508 | 0.2477 | 0.2509 | 0.3258 | 0.3571 | 0.3439 |
| 5 | 0.2226 | 0.2248 | 0.2254 | 0.2833 | 0.3181 | 0.3013 |
| 6 | 0.2039 | 0.2098 | 0.2087 | 0.2557 | 0.2935 | 0.2742 |
| 7 | 0.1908 | 0.1994 | 0.1971 | 0.2367 | 0.2773 | 0.2560 |
| 8 | 0.1812 | 0.1920 | 0.1888 | 0.2231 | 0.2661 | 0.2427 |
| 9 | 0.1740 | 0.1865 | 0.1828 | 0.2130 | 0.2582 | 0.2338 |
| 10 | 0.1685 | 0.1825 | 0.1782 | 0.2054 | 0.2526 | 0.2271 |
| 11 | 0.1642 | 0.1794 | 0.1747 | 0.1996 | 0.2487 | 0.2220 |
| 12 | 0.1608 | 0.1770 | 0.1720 | 0.1951 | 0.2461 | 0.2182 |
| 13 | 0.1580 | 0.1752 | 0.1699 | 0.1916 | 0.2447 | 0.2153 |
| 14 | 0.1559 | 0.1738 | 0.1683 | 0.1890 | 0.2443 | 0.2132 |
| 15 | 0.1541 | 0.1727 | 0.1670 | 0.1871 | 0.2448 | 0.2118 |
| 16 | 0.1527 | 0.1718 | 0.1660 | 0.1857 | 0.2461 | 0.2109 |
| Median | 0.2088 | 0.2176 | 0.2165 | 0.2631 | 0.3125 | 0.2879 |
| SE | 0.0318 | 0.0255 | 0.0291 | 0.0480 | 0.0473 | 0.0516 |

*Table 13. Depletion-based stochastic stock reduction analysis model parameters and fisheries management reference quantities (mean, standard deviation (SD) and quantiles (including median estimates (50%)): overfishing limit in 2000 ($OFL_{2000}$), depletion-corrected average catch (DCAC), carrying capacity (K), abundance-based MSY ($N_{MSY}$), maximum sustainable yield (MSY), natural mortality (M), delta, fishing mortality at MSY ($F_{MSY}$) and exploitation rate at MSY ($E_{MSY}$), for Rat River Dolly Varden. Depletion-based stochastic stock reduction analysis model parameters are detailed in Table 3.*

| Parameter | Mean | SD | 2.50% | 25% | 50% | 75% | 97.50% |
|---|---|---|---|---|---|---|---|
| $OFL_{2000}$ | 1,305 | 612 | 481 | 879 | 1,190 | 1,594 | 2,836 |
| DCAC | 1,129 | 183 | 730 | 1,011 | 1,144 | 1,262 | 1,437 |
| *K* | 19,542 | 5,790 | 10,499 | 15,442 | 18,804 | 22,764 | 32,769 |
| $N_{MSY}$ | 7,778 | 2,357 | 4,108 | 6,131 | 7,466 | 9,090 | 13,255 |
| MSY | 1,247 | 312 | 722 | 1,034 | 1,217 | 1,419 | 1,971 |
| *M* | 0.2901 | 0.1275 | 0.1157 | 0.1991 | 0.2653 | 0.3556 | 0.6050 |
| Delta | 0.5999 | 0.0977 | 0.4019 | 0.5331 | 0.6026 | 0.6694 | 0.7792 |
| $F_{MSY}$ | 0.2320 | 0.1135 | 0.0823 | 0.1525 | 0.2087 | 0.2863 | 0.5129 |
| $E_{MSY}$ | 0.1737 | 0.0627 | 0.0742 | 0.1277 | 0.1658 | 0.2120 | 0.3144 |

*Table 14. Surplus production model parameters and fisheries management reference quantities (mean, standard deviation (SD) and quantiles (including median estimates (50%)), estimated either by maximum likelihood estimates (MLE) or Markov chain Monte Carlo (MCMC) for Rat River Dolly Varden. Model parameters are detailed in Table 4.*

| Variable | Mean | SD | 2.50% | 25% | 50% | 75% | 97.50% |
|---|---|---|---|---|---|---|---|
| MLE | | | | | | | |
| log(K) | 10.0030 | 0.0139 | - | - | - | - | - |
| log(q) | -7.6374 | 0.1055 | - | - | - | - | - |
| log(σ) | 4.2767 | 0.2694 | - | - | - | - | - |
| log(τ) | 2.4987 | 0.1489 | - | - | - | - | - |
| *K* | 22,093 | 307 | - | - | - | - | - |
| *r* | 0.1819 | 0.0287 | - | - | - | - | - |
| q | 4.82E-04 | - | - | - | - | - | - |
| *σ* | 0.1178 | - | - | - | - | - | - |
| *τ* | 0.2867 | - | - | - | - | - | - |
| $N_{MSY}$ | 11,047 | - | - | - | - | - | - |
| MSY | 1,005 | - | - | - | - | - | - |
| $F_{MSY}$ | 0.0909 | - | - | - | - | - | - |
| $E_{MSY}$ | 0.2401 | - | - | - | - | - | - |
| MCMC | | | | | | | |
| $N_{MSY}$ | 16,273 | 7,042 | 11,094 | 12,014 | 13,689 | 17,403 | 38,293 |
| MSY | 2,653 | 1,385 | 1,288 | 1,812 | 2,242 | 2,969 | 6,788 |
| $F_{MSY}$ | 0.1610 | 0.0315 | 0.1058 | 0.1384 | 0.1584 | 0.1815 | 0.2302 |
| $N/N_{MSY}$ | 1.8952 | 0.0686 | 1.7072 | 1.8698 | 1.9127 | 1.9402 | 1.9756 |
| $F/F_{MSY}$ | 0.0932 | 0.0373 | 0.0289 | 0.0676 | 0.0910 | 0.1150 | 0.1764 |
| $E_{MSY}$ | 0.1293 | 0.0234 | 0.0875 | 0.1126 | 0.1278 | 0.1448 | 0.1797 |

*Table 15. Median and standard deviation (SD) for Rat River Dolly Varden population dynamics model parameters from Depletion-based stock reduction analysis (DB-SRA), surplus production (SPM), and integrated statistical catch-at-age (iSCAM) models: maximum sustainable yield (MSY), abundance and biomass at MSY ($N_{MSY}$, $B_{MSY}$), fishing mortality at MSY ($F_{MSY}$), natural mortality (M), and exploitation rate at MSY ($E_{MSY}$)). iSCAM also provides the estimates for stock spawning biomass (SSB). An average for these parameters is calculated using weighted inverse variance (WIV).*

| Parameter | | DB-SRA | | SPM | | iSCAM | | WIV | |
|---|---|---|---|---|---|---|---|---|---|
| | | Median | SD | Median | SD | Median | SD | Median | SD |
| Abundance | $N_{MSY}$ | 7,466 | 2,357 | 13,689 | 7,042 | 23,968 | 13,003 | 10,813 | 1,555 |
| | MSY | 1,217 | 312 | 2,242 | 1,385 | 1,007 | 715 | 1,301 | 188 |
| | $F_{MSY}$ | 0.2087 | 0.1135 | 0.1584 | 0.0315 | 0.2546 | 0.1167 | 0.1842 | 0.0204 |
| | $M$ | 0.2653 | 0.1275 | 0.2750 | 0.2020 | 0.2906 | 0.0144 | 0.2873 | 0.0121 |
| | $E_{MSY}$ | 0.1658 | 0.0627 | 0.1278 | 0.0234 | 0.1963 | 0.0777 | 0.1486 | 0.0140 |
| Biomass | $B_{MSY}$ | - | - | - | - | 6,520 | 1,293 | 6,520 | 1,293 |
| | MSY | - | - | - | - | 1,660 | 200 | 1,660 | 200 |
| | $F_{MSY}$ | - | - | - | - | 0.2546 | 0.1167 | 0.2546 | 0.1167 |
| | $M$ | - | - | - | - | 0.2906 | 0.0144 | 0.2906 | 0.0144 |
| | $E_{MSY}$ | - | - | - | - | 0.1963 | 0.0777 | 0.1963 | 0.0777 |
| SSB | $SSB_{MSY}$ | - | - | - | - | 642.94 | 309.66 | 642.94 | 309.66 |
| | MSY | - | - | - | - | 132.40 | 105.15 | 132.40 | 105.15 |
| | $F_{MSY}$ | - | - | - | - | 0.2546 | 0.1167 | 0.2546 | 0.1167 |
| | $M$ | - | - | - | - | 0.2906 | 0.0144 | 0.2906 | 0.0144 |
| | $E_{MSY}$ | - | - | - | - | 0.1963 | 0.0777 | 0.1963 | 0.0777 |

Note: DB-SRA and SPM estimate the fishable portion of the population, which corresponds to fish > 440 mm for Rat River Dolly Varden, while iSCAM estimates the age-specific population abundance and biomass.

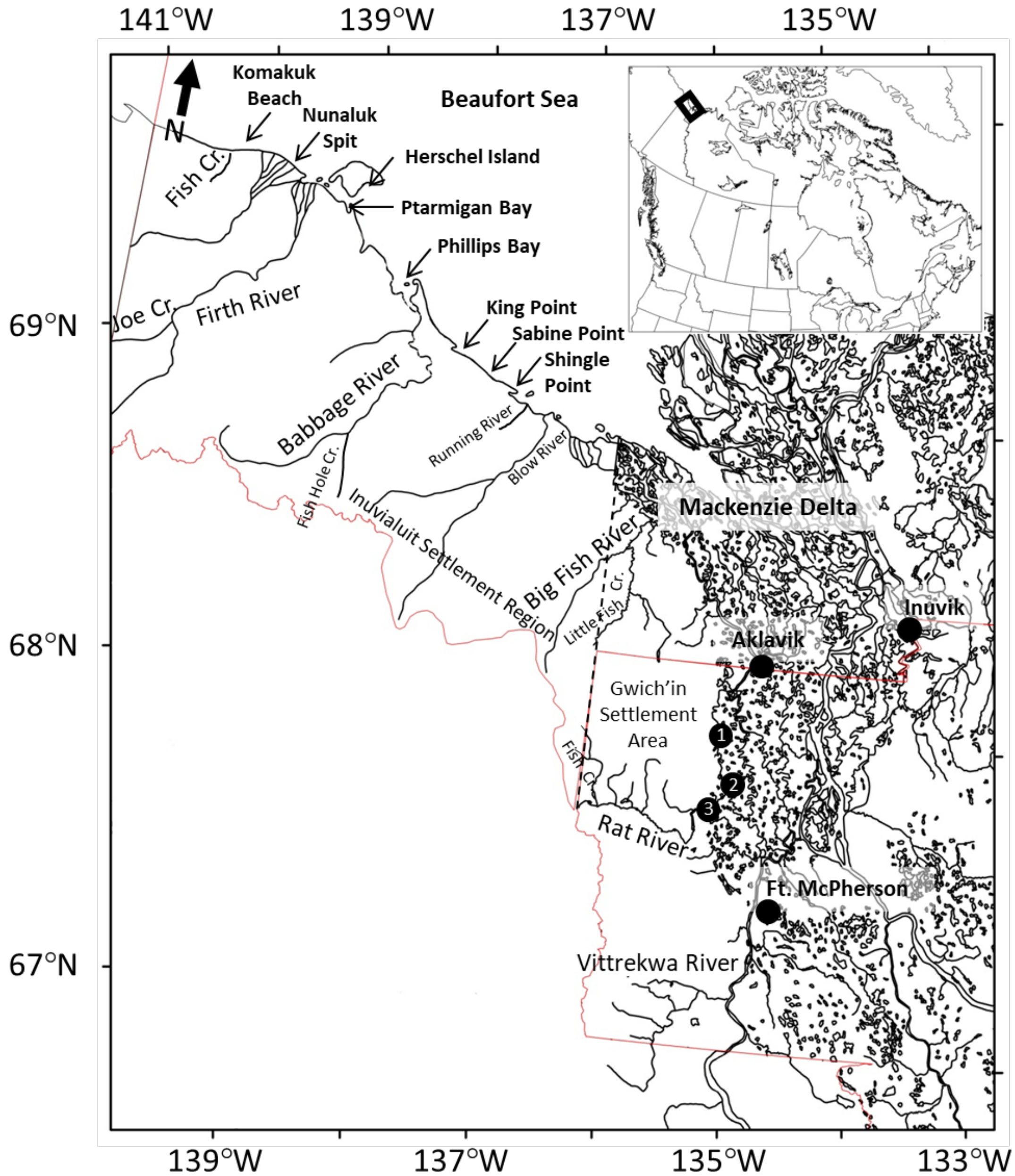


*Figure 1. Locations in the Inuvialuit Settlement Region and Gwich'in Settlement Area (red lines delineate borders) within the Northwest Territories, Canada, where community-based monitoring and biological sampling of Rat River Dolly Varden occurred from early May to mid-September: Big Eddy in the Mackenzie Delta (Husky Channel) (1), the Rat River mouth (2), and the Rat River at Destruction City (3).*

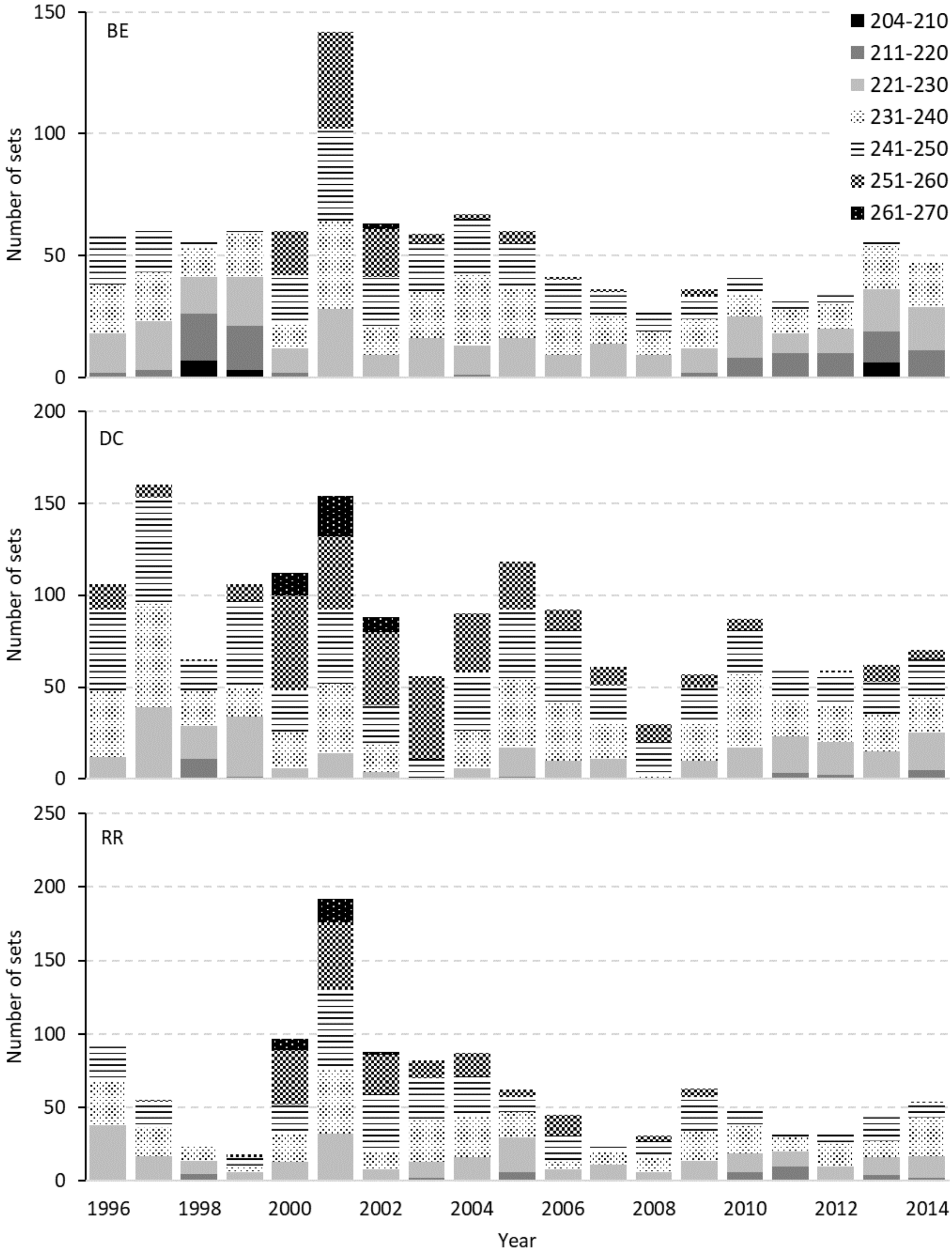


*Figure 2. Set frequency grouped by day of the year at Big Eddy (BE), Destruction City (DC), and the mouth of the Rat River (RR), 1996–2014.*

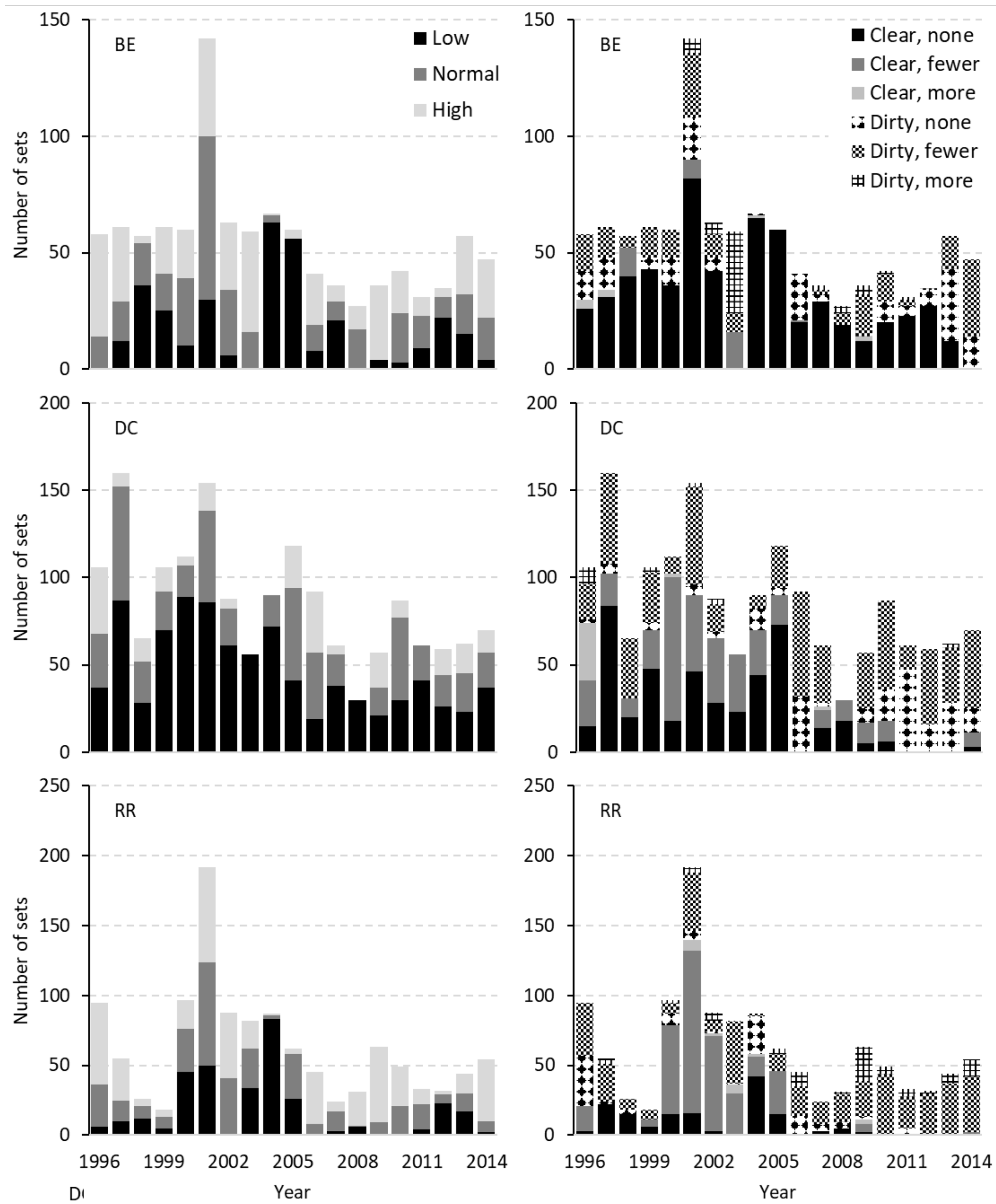


*Figure 3. Set frequency grouped by water level (left panel) and turbidity (right panel) at Big Eddy (BE), Destruction City (DC) and the mouth of the Rat River (RR), 1996–2014.*

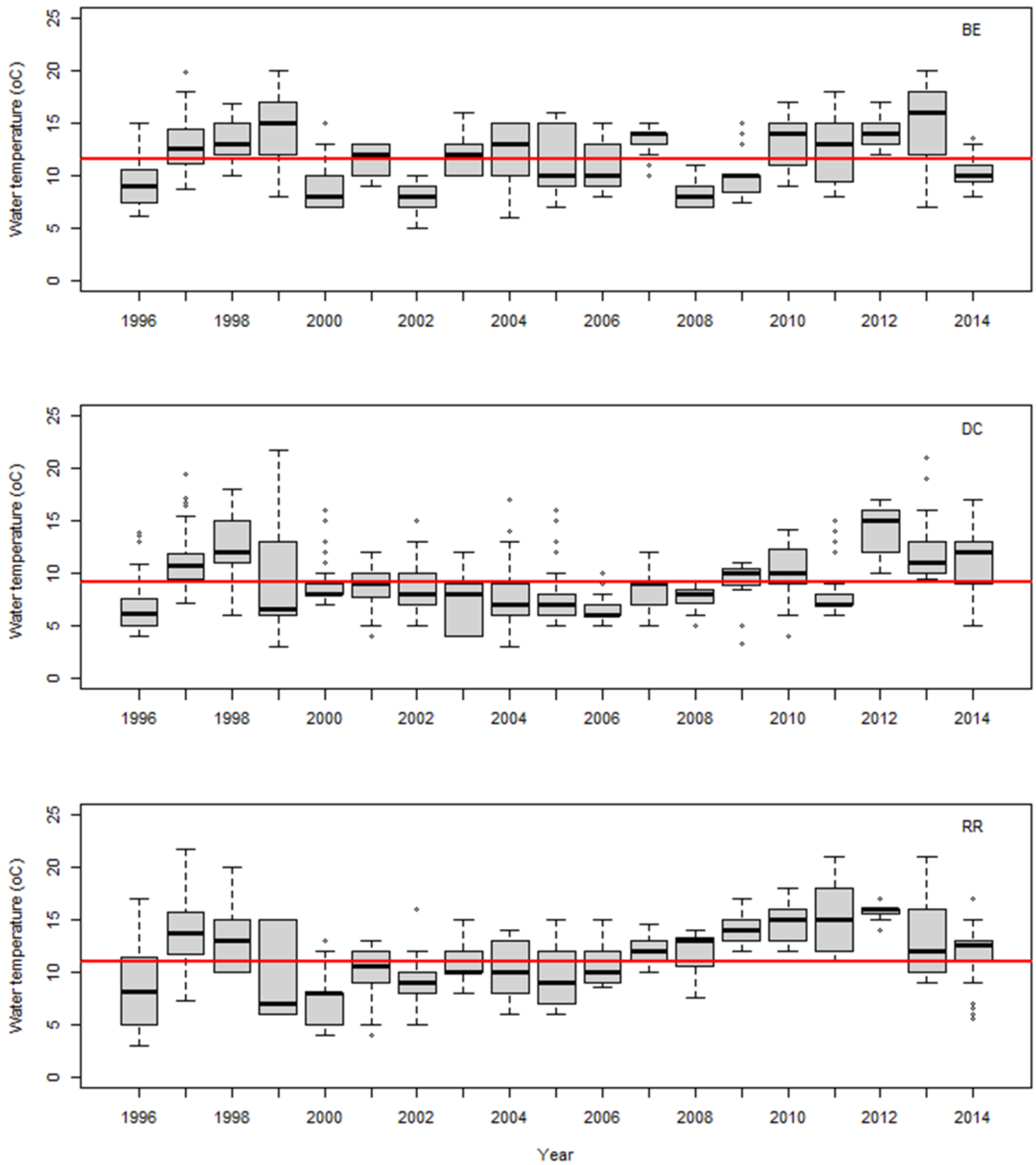


*Figure 4. Water temperature, presented in the form of box-plots (black bar = median, grey box = first and third quartiles, and dashed line and circles = range in values) recorded by fisheries monitors at Big Eddy (BE), Destruction City (DC), and the mouth of the Rat River (RR), 1996–2014. The red line indicates medians throughout the entire time series.*

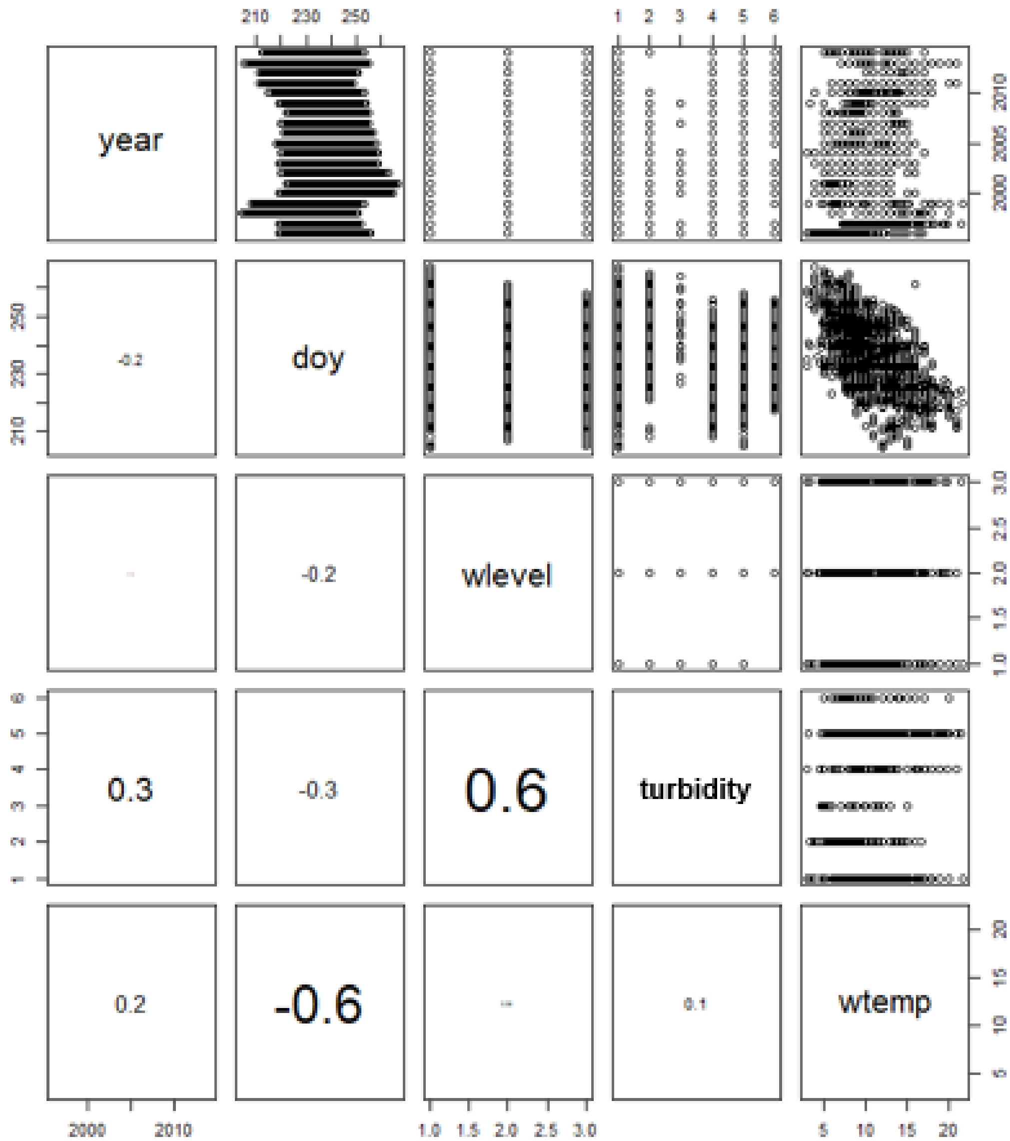


*Figure 5. Pairwise correlations to test the presence of collinearity between independent variables, year, day of the year (doy), water level (wlevel), turbidity, and water temperature (wtemp), which were considered as candidate variables to standardize Rat River Dolly Varden CPUE.*

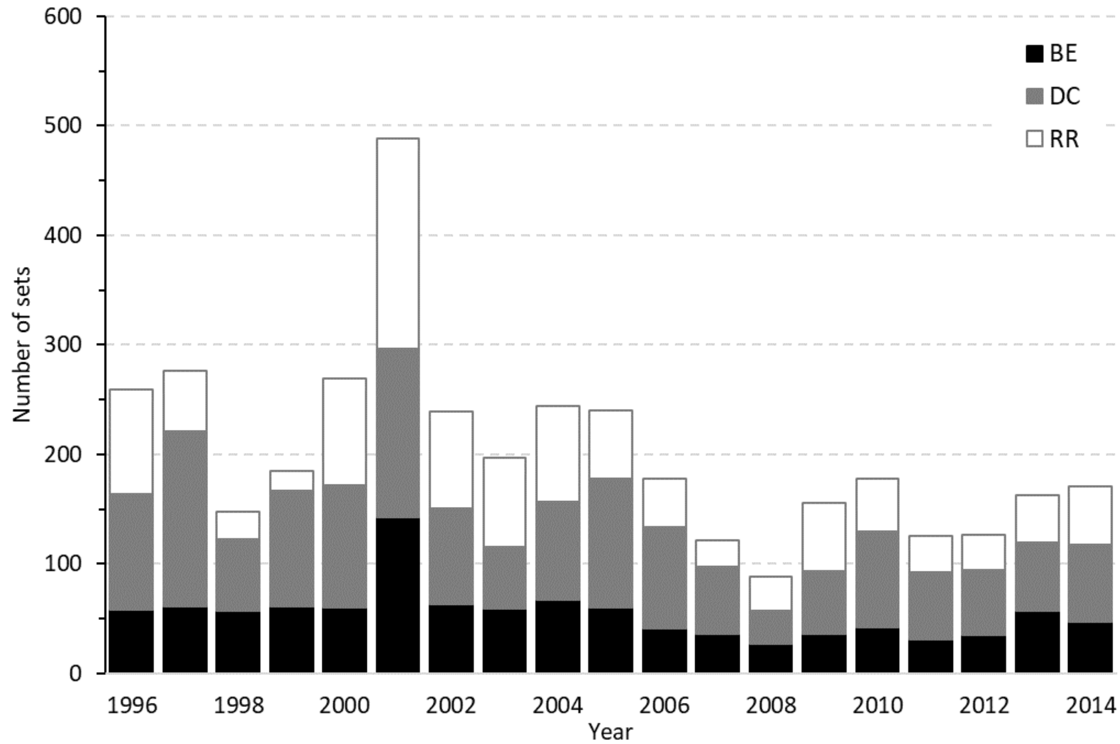


*Figure 6. Number of gillnet sets during the Rat River Harvest Monitoring Program at Big Eddy (BE), the mouth of Rat River (RR), and Destruction City (DC),1996–2014.*

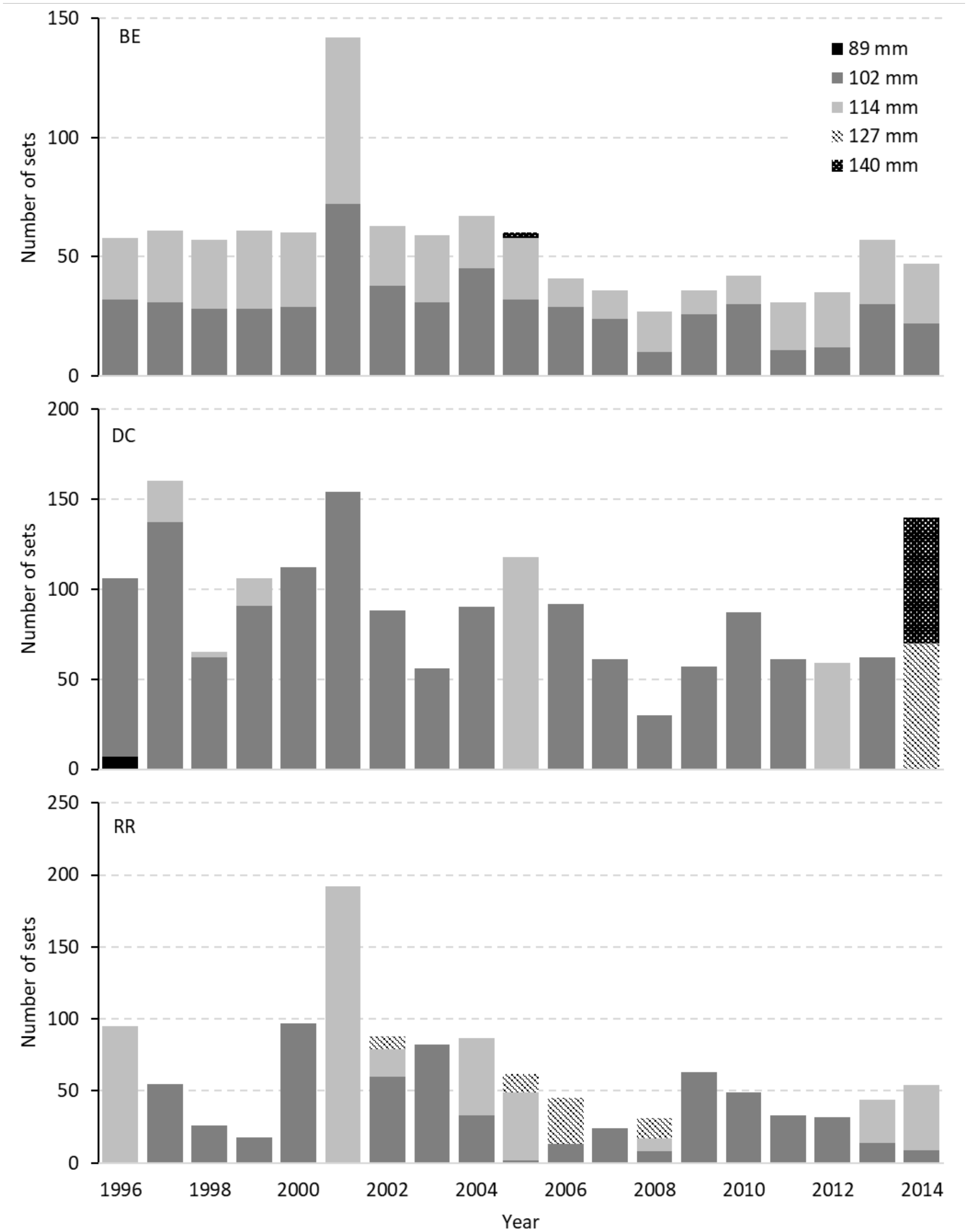


*Figure 7. Set frequency by gillnet mesh size (mm) used by fisheries monitors at Big Eddy (BE), Destruction City (DC), and the mouth of Rat River (RR), 1996–2014.*

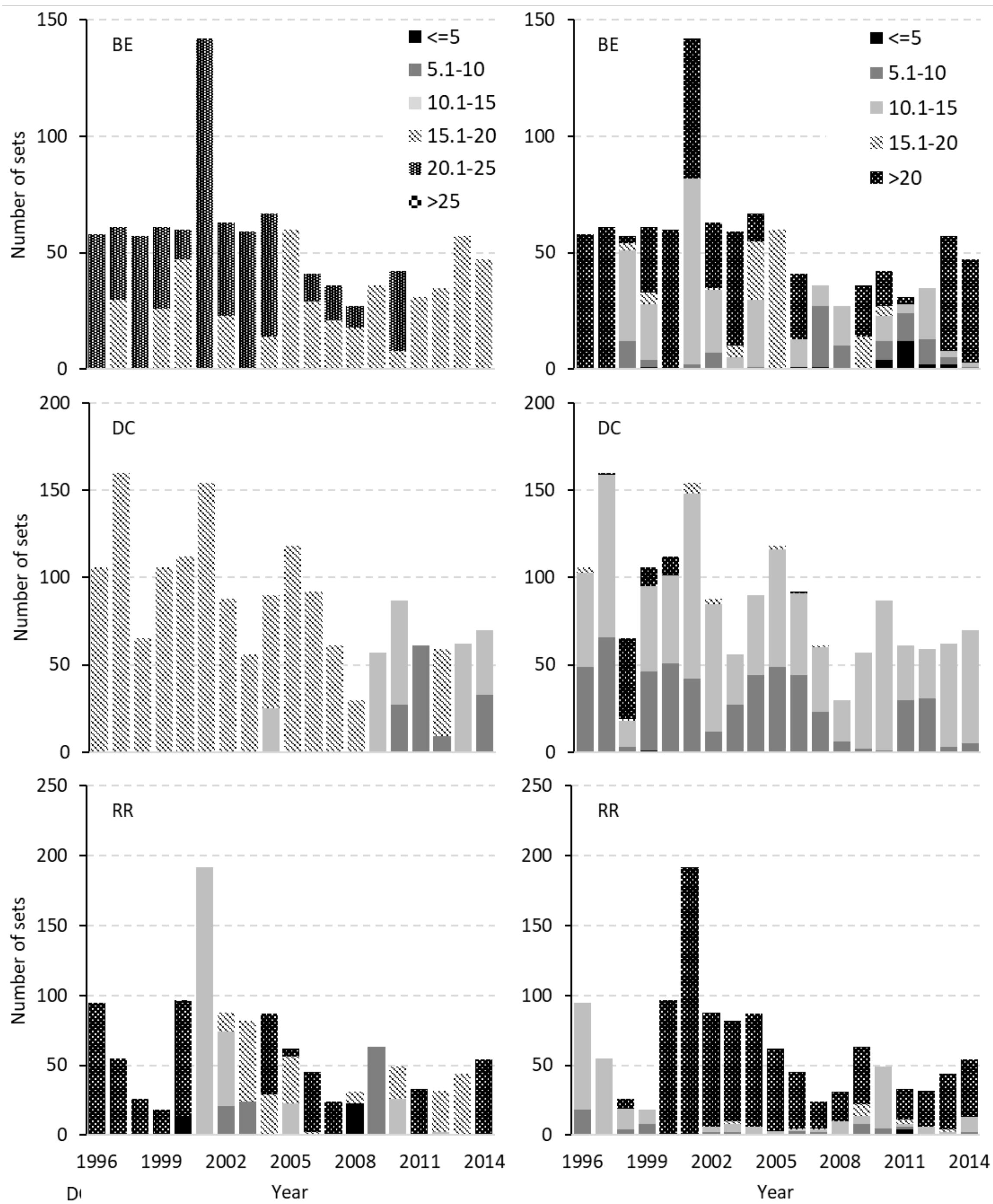


*Figure 8. Set frequency grouped by gillnet panel length (left column) and soak hour classes (right column) at Big Eddy (BE), Destruction City (DC), and the mouth of Rat River (RR),1996–2014.*

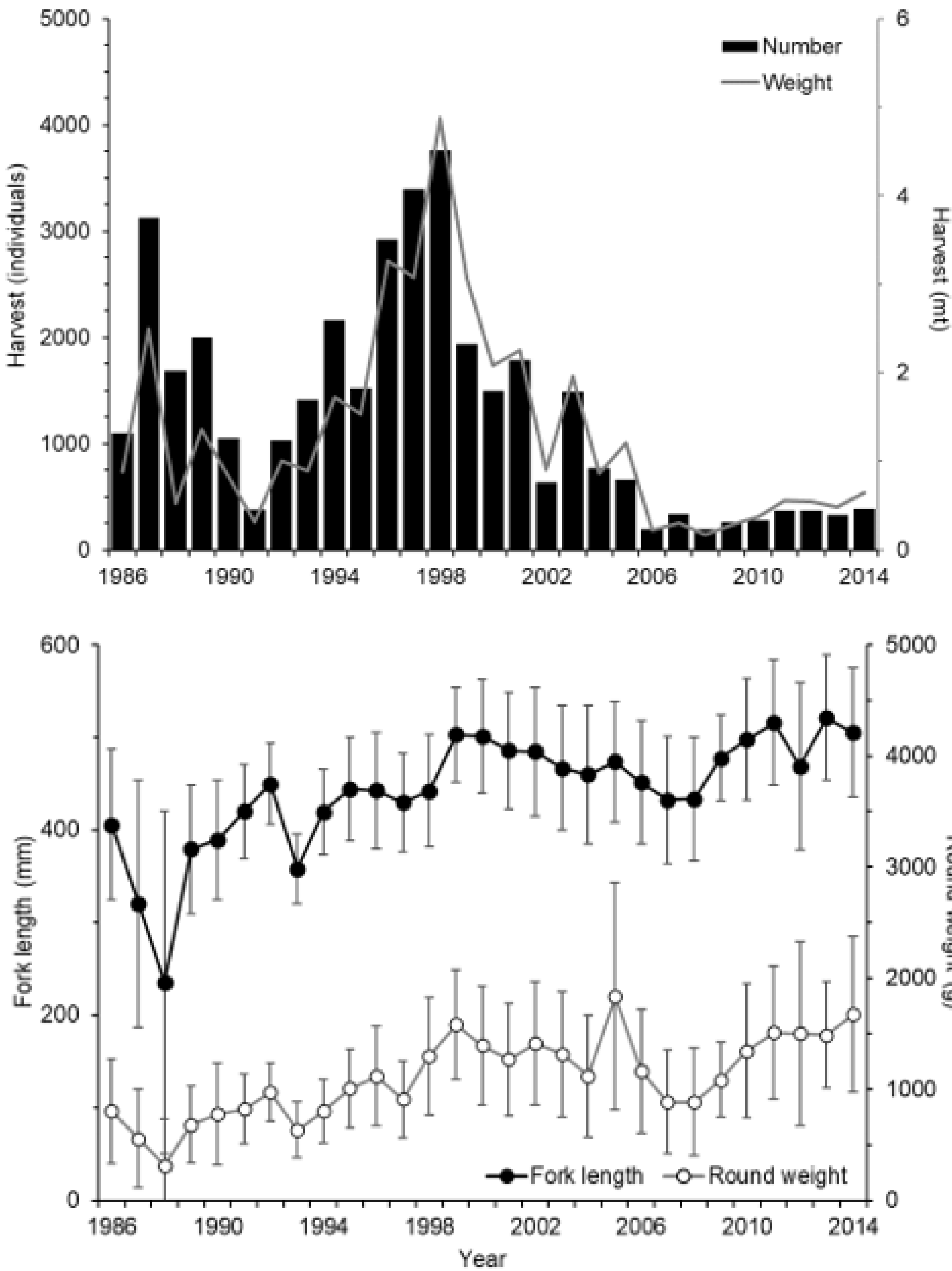


*Figure 9. Subsistence harvest number (black bar) and weight (metric tonne, grey line) estimates (upper panel) as well as average size (lower panel, mean ± SD) for Rat River Dolly Varden, 1986–2014. A significant decline in the harvest led to the voluntary closure of the subsistence fishery during 2006–2008. Since 2009, the fisheries resumed with an annual harvest of under 500 fish.*

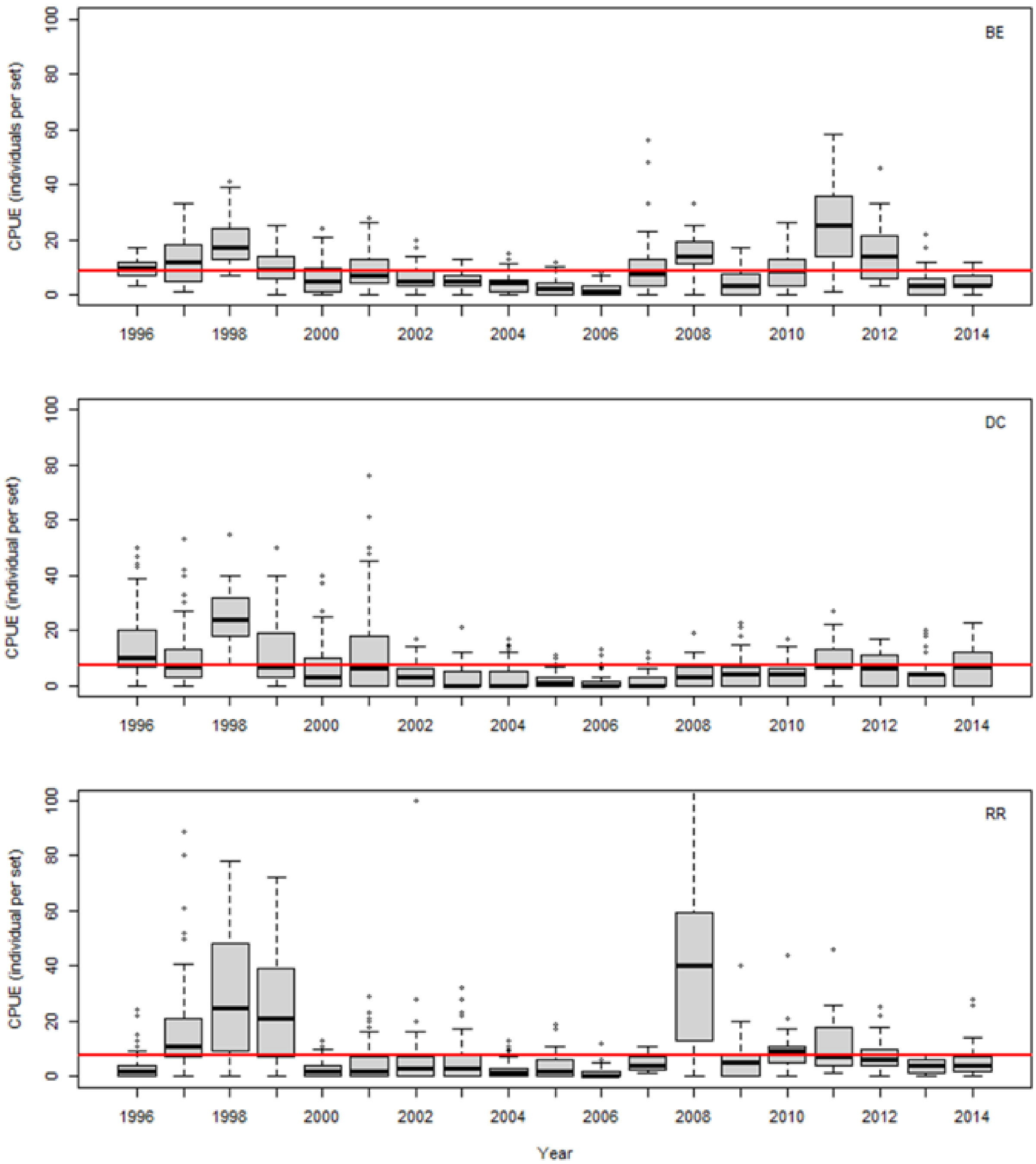


*Figure 10. Nominal catch per unit effort (CPUE), presented in the form of box-plots (black bar = median, grey box = the first and third quartiles, dashed line and circles = range in values) for Rat River Dolly Varden harvested at Big Eddy (BE), Destruction City (DC), and the mouth of Rat River (RR), 1996–2014. The red line indicates the medians throughout entire time series.*

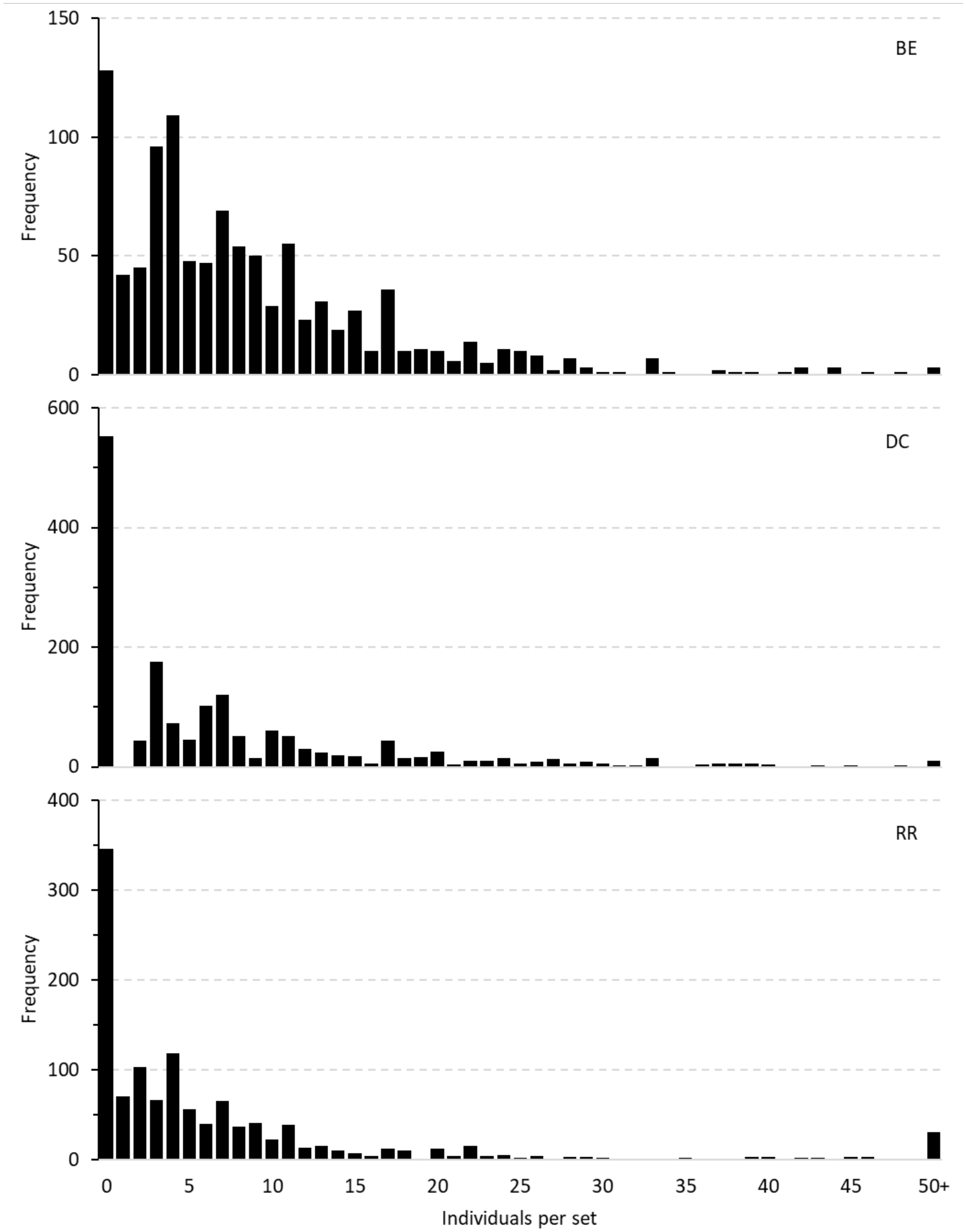


*Figure 11. Frequency of nominal CPUE for Rat River Dolly Varden harvested by subsistence fisheries monitors at Big Eddy (BE), Destruction City (DC) and the mouth of Rat River (RR), 1996–2014.*

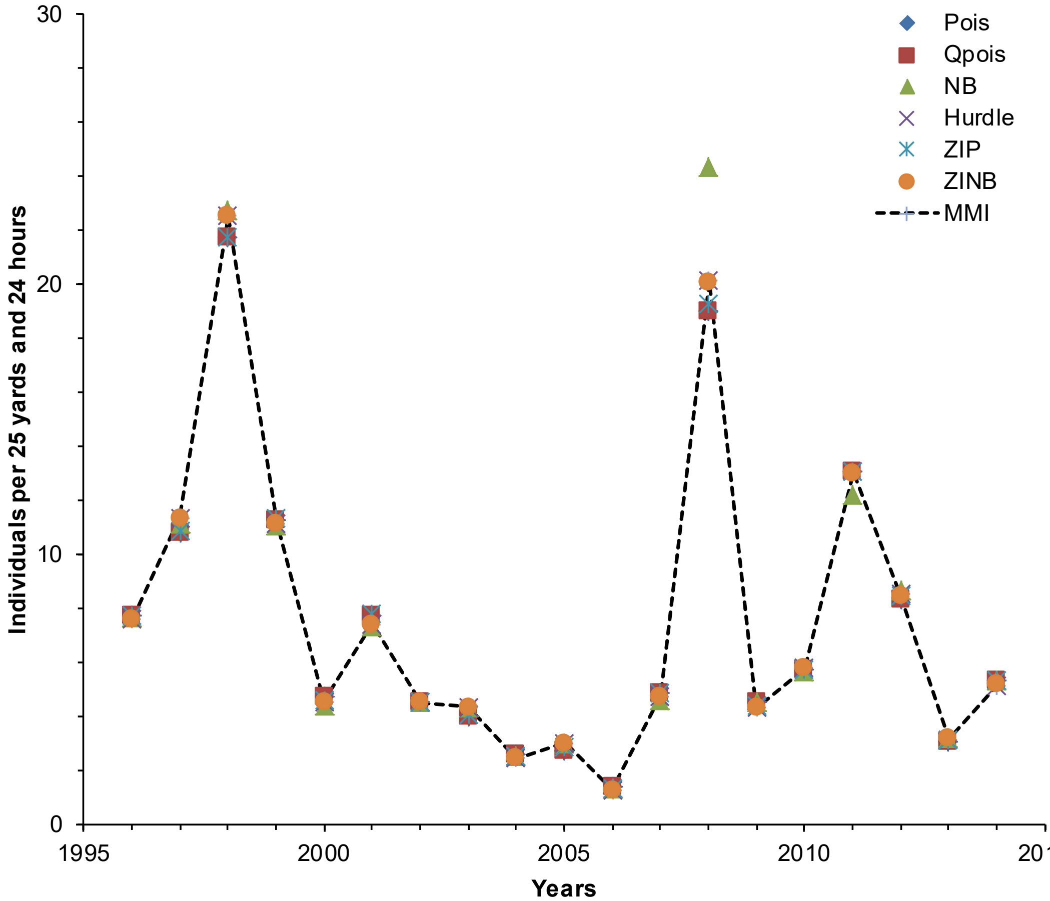


*Figure 12. Catch-per-unit effort (number of fish per 25 yards and 24 hours) standardization by means of generalized linear models (Poisson (Pois), quasi-Poisson (Qpois) and negative binomial (NB)), zero-augmented models (Hurdle, zero-inflated Poisson (ZIP) and Zero-inflated negative binomial (ZINB)), as well as from the multimodel inference (MMI) for Rat River Dolly Varden, 1996–2014.*

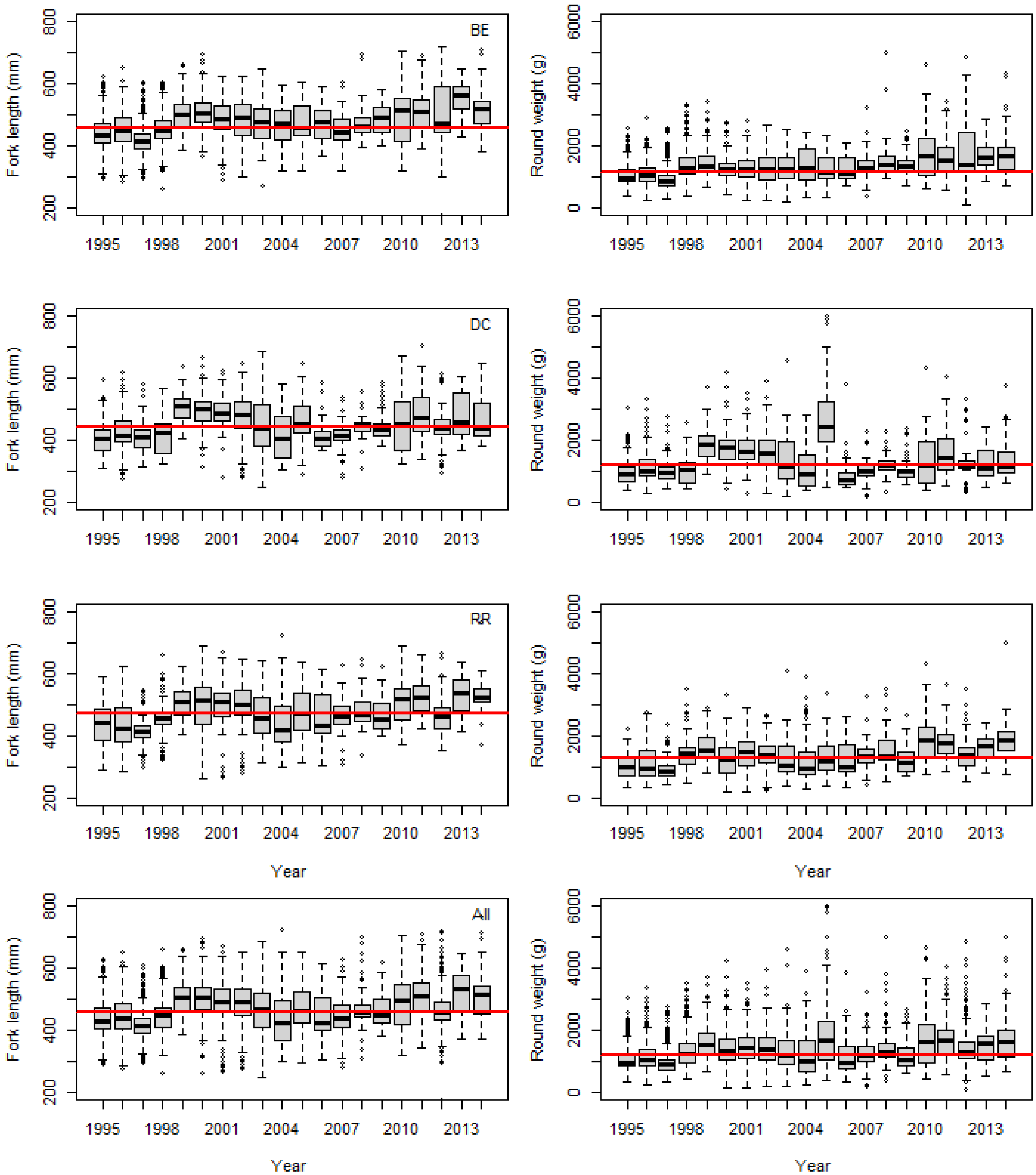


*Figure 13. Box-plots ((black bar = median, grey box = the first and third quartiles, dashed line and circles = range in values) for fork length (left panel) and round weight (right panel) of Rat River Dolly Varden harvested at Big Eddy (BE), Destruction City (DC) and the mouth of the Rat River (RR), 1995–2014. The red line indicates the medians throughout the entire time series.*

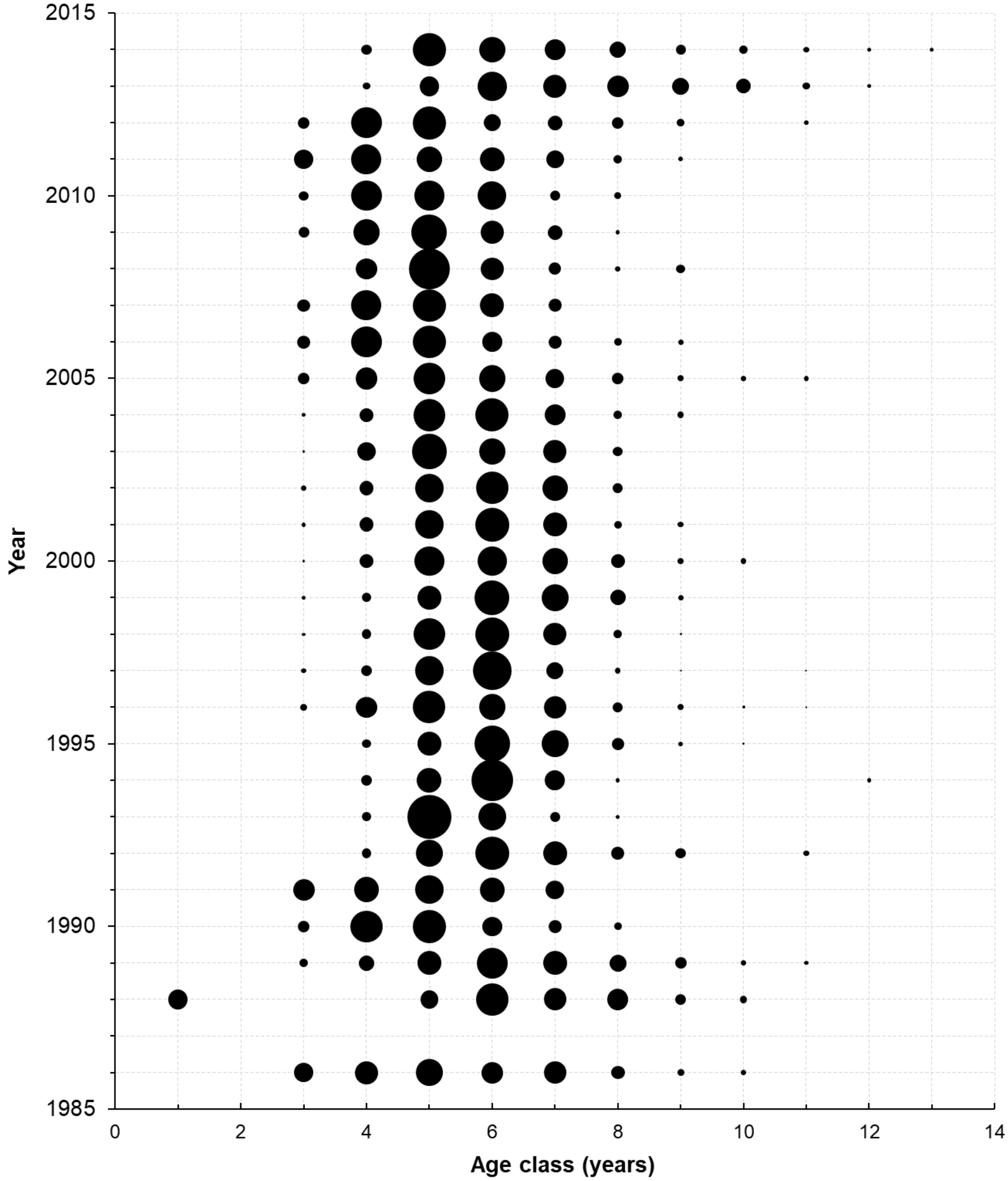


*Figure 14. Proportion-at-age for Rat River Dolly Varden, 1986–2014.*

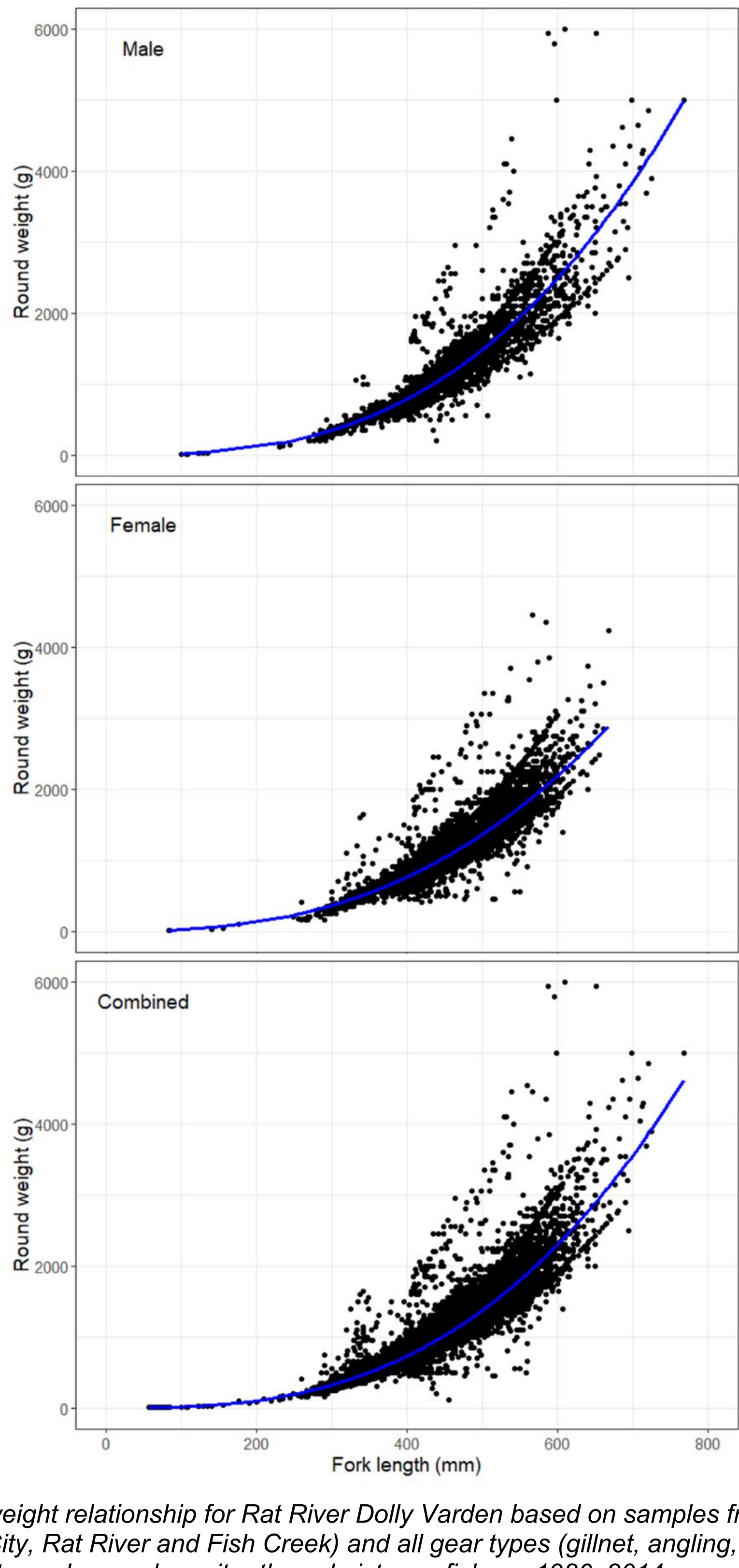


*Figure 15. Length-weight relationship for Rat River Dolly Varden based on samples from all areas (Big Eddy, Destruction City, Rat River and Fish Creek) and all gear types (gillnet, angling, electroshock, seine, and hoopnet) used to explore and monitor the subsistence fishery, 1986–2014.*

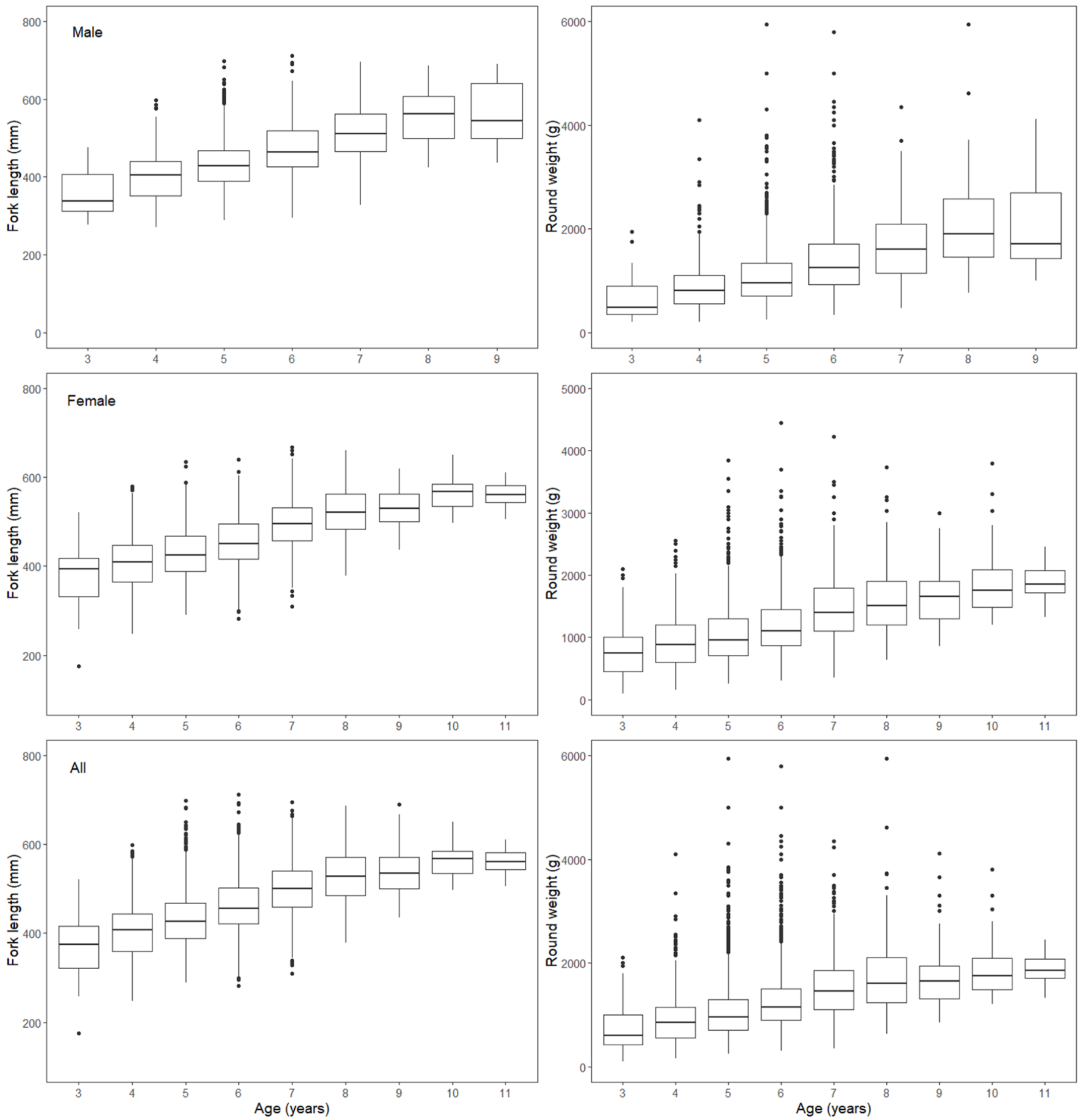


*Figure 16. Growth of fork length (mm) (left panels) and round weight (g) (right panels) at age (years) for Rat River Dolly Varden, sampled by all gear types (gillnet, angling, electroshock, seine, and hoopnet) in the subsistence harvest areas (Big Eddy, Destruction City, Rat River and Fish Creek), 1986–2014.*

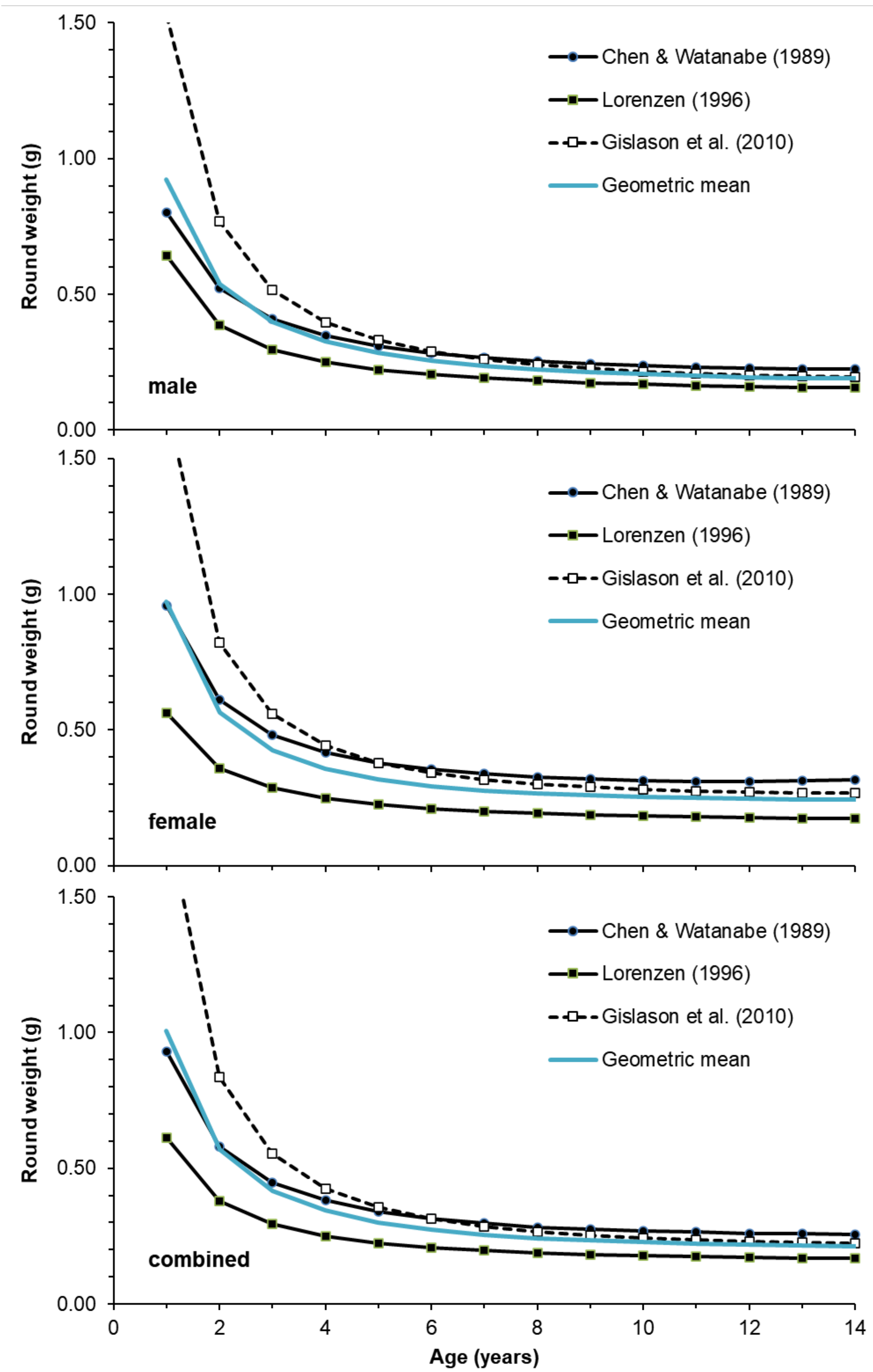


*Figure 17. Natural mortality estimates from multiple life-history models and the geometric mean for Rat River Dolly Varden.*

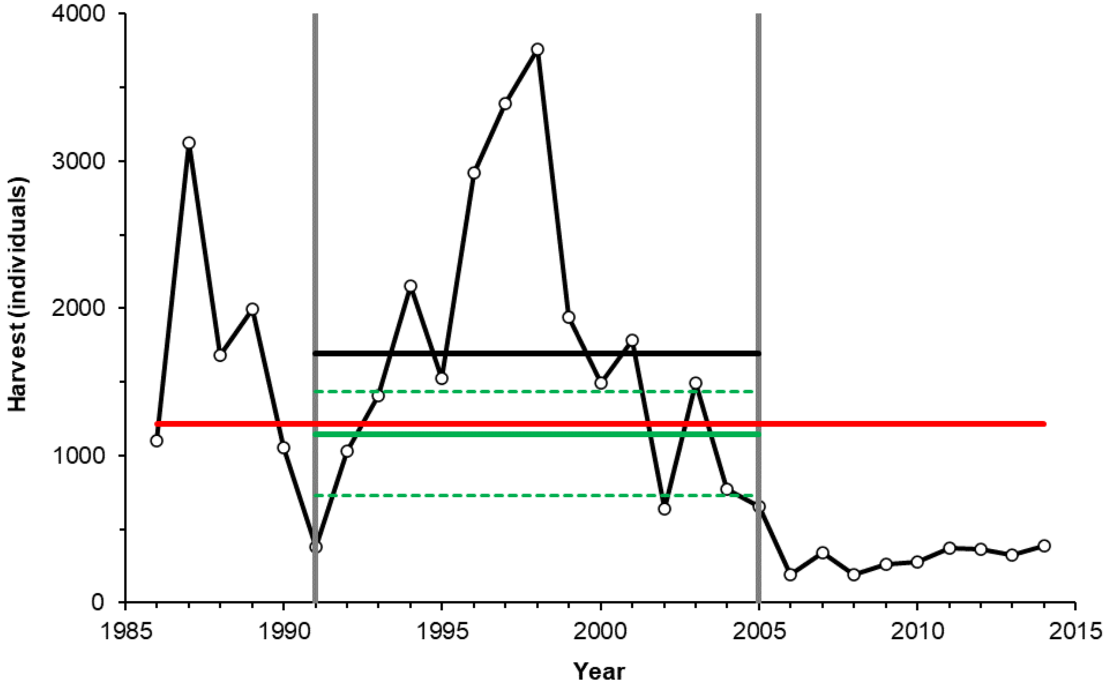


*Figure 18. Depletion-based stock reduction analysis for Rat River Dolly Varden subsistence harvest, 1986–2014. Maximum sustainable yield (number of individual fish) is expressed by a red line. Mean depletion-corrected average catch (solid green line with 95% confidential intervals (dotted green lines)) and average subsistence harvest (black line) for 1991–2005 (bracketed by vertical grey bars).*

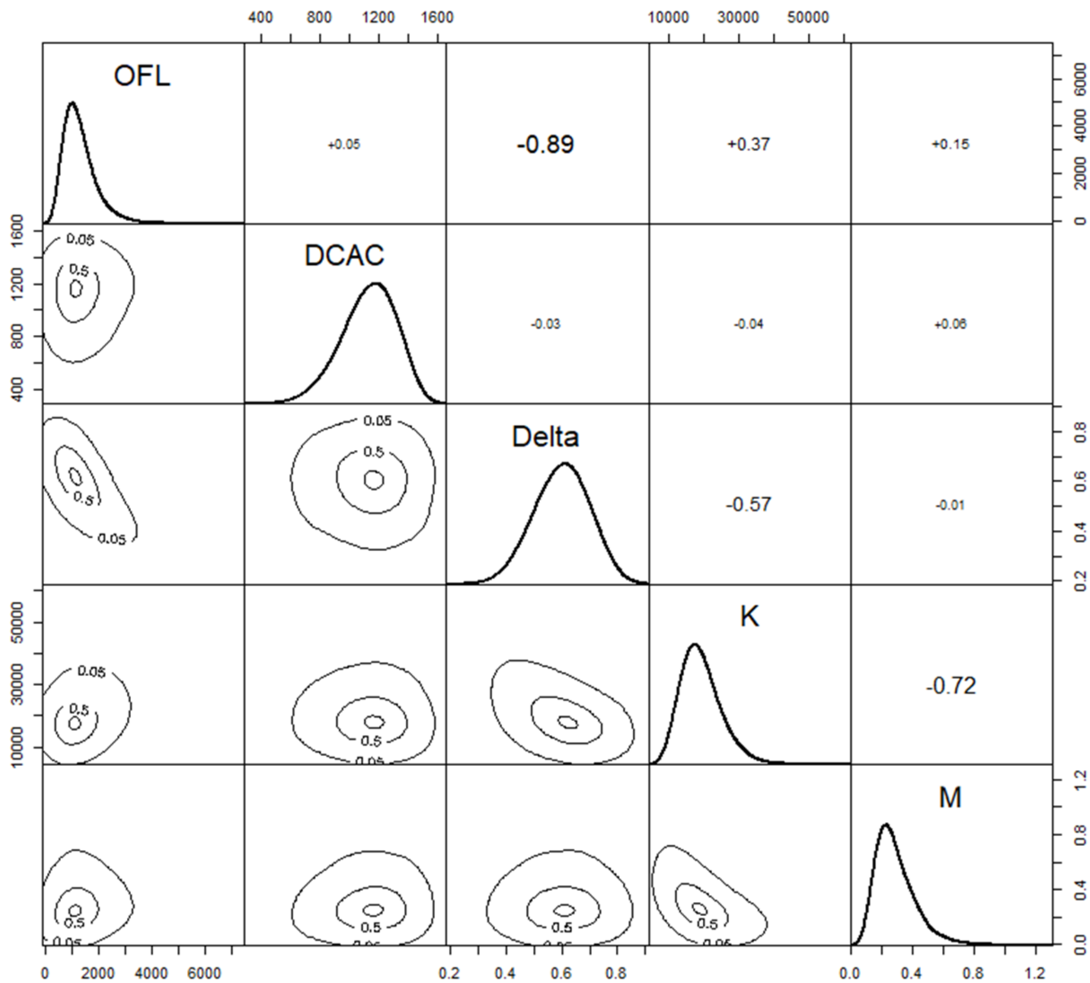


*Figure 19. Isograms for depletion-based stock reduction analysis model parameters including overfishing limit (OFL), depletion-corrected average catch (DCAC), relative stock status (delta), carrying capacity (K), and natural mortality (M) for Rat River Dolly Varden. Isograms (to the left of the diagonal) indicated covariate relations and values in boxes (to the right of the diagonal) are Spearman correlation coefficients. A significant negative correlation between OFL and delta is indicated using bold font.*

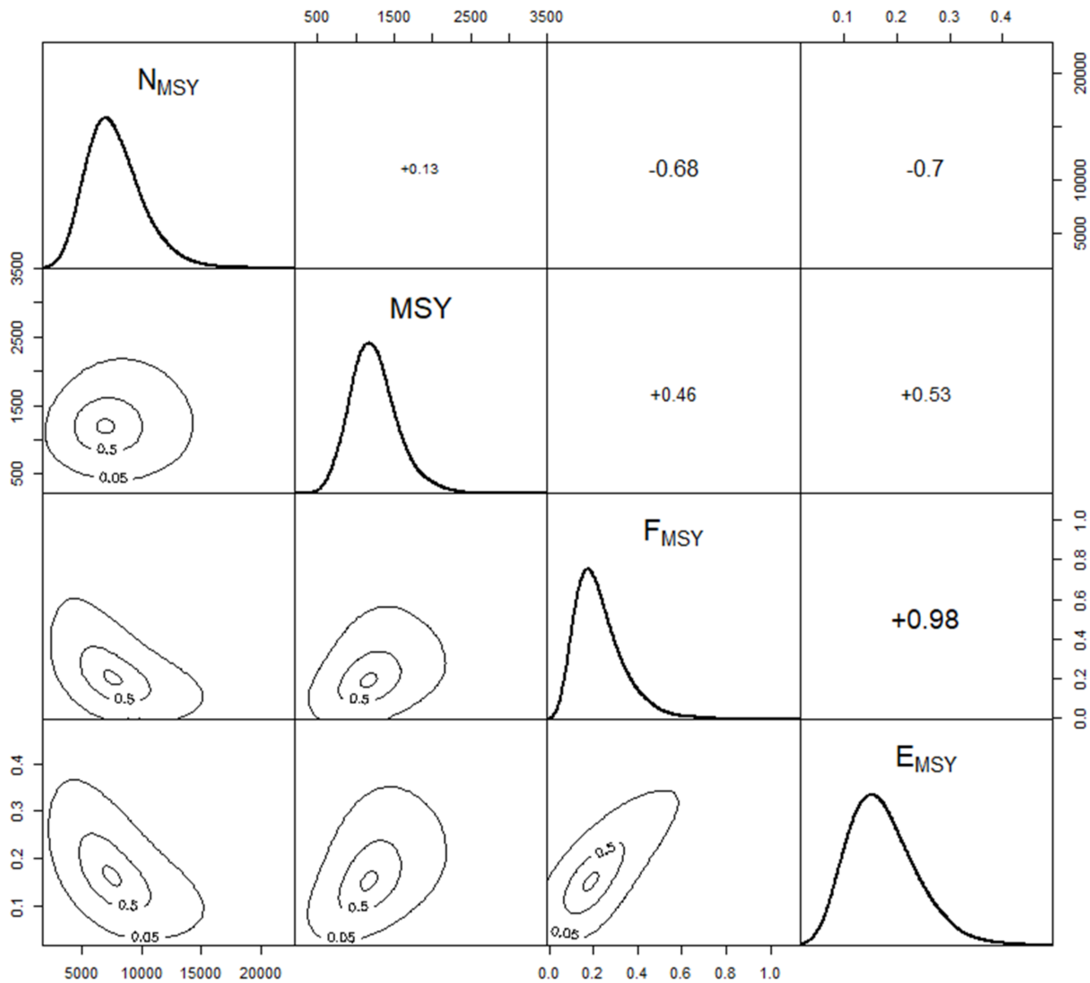


*Figure 20. Isograms of fisheries population dynamics parameters including maximum sustainable yield (MSY), abundance at MSY ($N_{MSY}$), fishing mortality and exploitation at MSY ($F_{MSY}$, and $E_{MSY}$), estimated by depletion-based stock reduction analysis for Rat River Dolly Varden. Isograms (to left of diagonal) indicated covariate relations and values in boxes (to right of diagonal) are Spearman correlation coefficients. A significant positive correlation between $F_{MSY}$ and $E_{MSY}$ is indicated using bold font.*

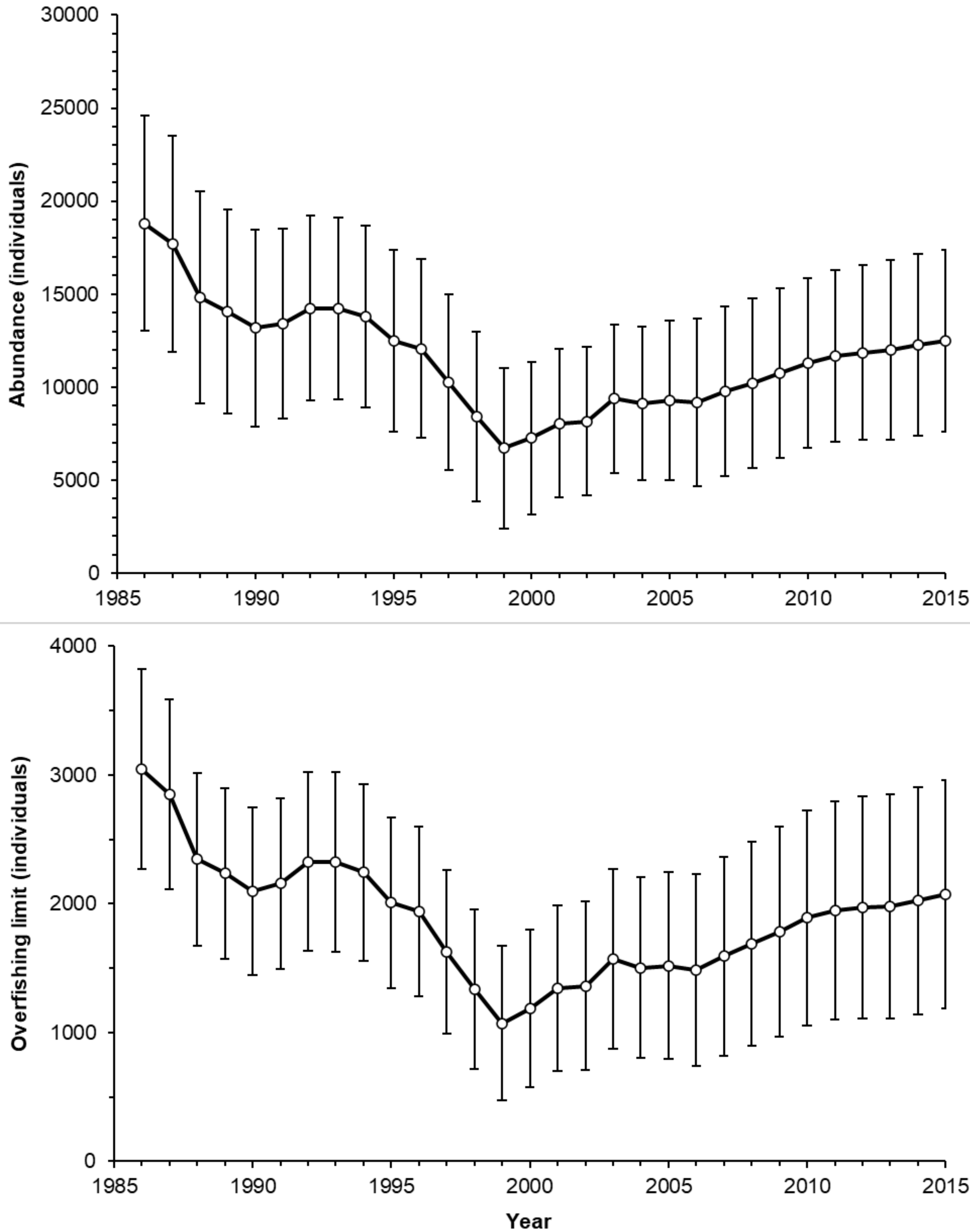


*Figure 21. Population abundance (top) and overfishing limit (OFL) (bottom) expressed as number of fish (median ± SD) for Rat River Dolly Varden, 1986–2015, estimated by depletion-based stock reduction analysis.*

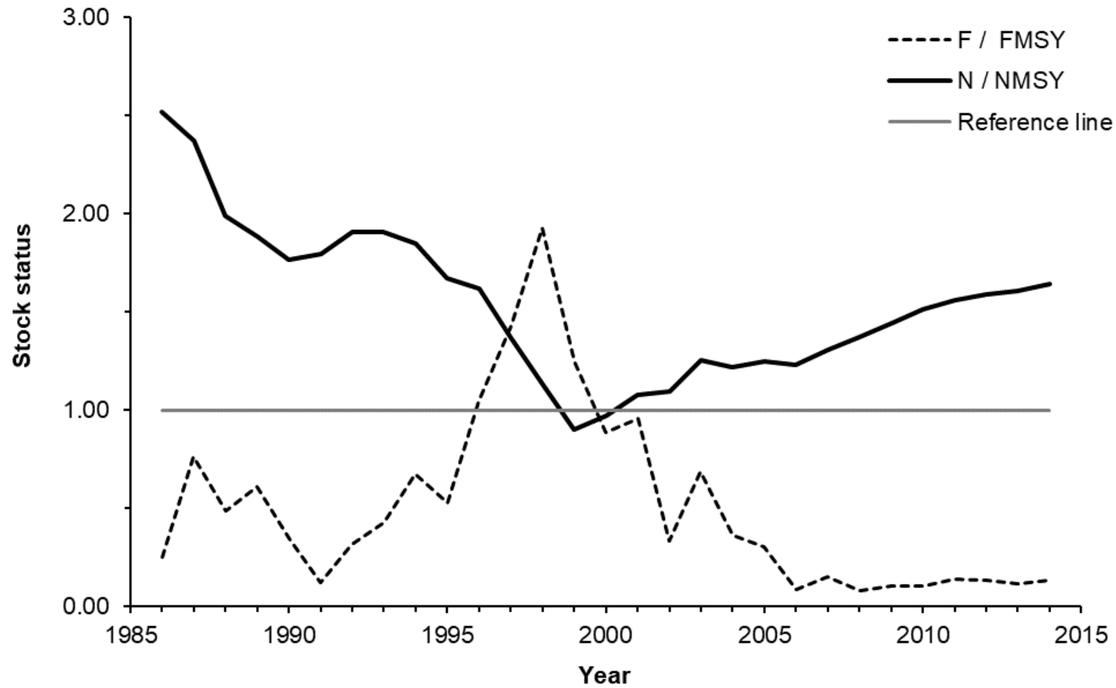


*Figure 22. Abundance ($N/N_{MSY}$) and exploitation ($F/F_{MSY}$) statuses from depletion-based stock reduction analysis for Rat River Dolly Varden, 1986–2014. The grey line is the stock status reference when $N/N_{MSY}$ or $F/F_{MSY}$ = 1.*

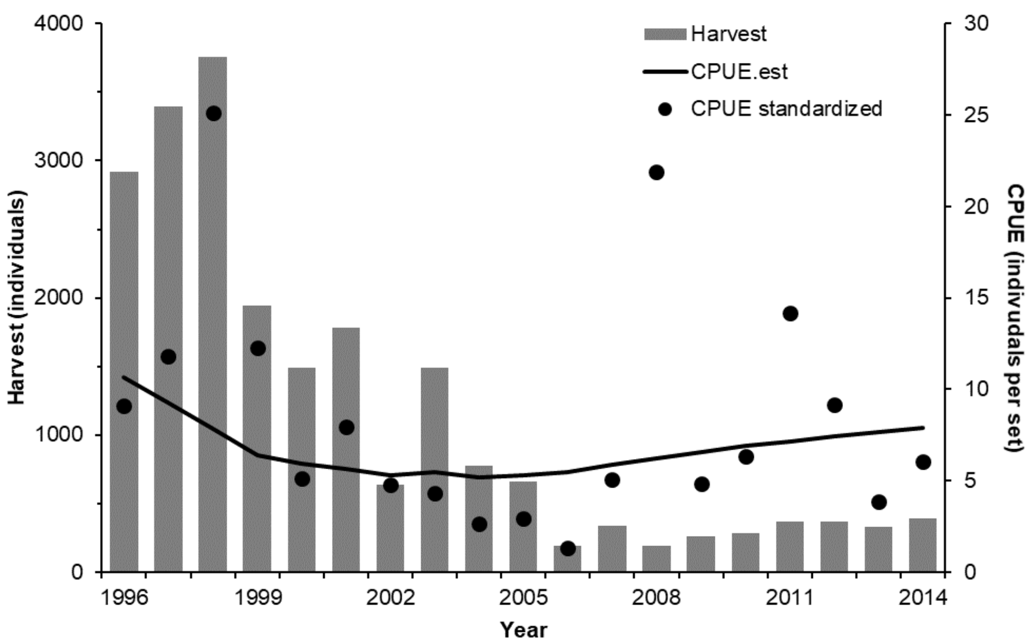


*Figure 23. Subsistence harvest (number of fish (grey bars), standardized (black dots), and estimated catch-per-unit effort (black line) from the Rat River Dolly Varden surplus production model, 1996–2014.*

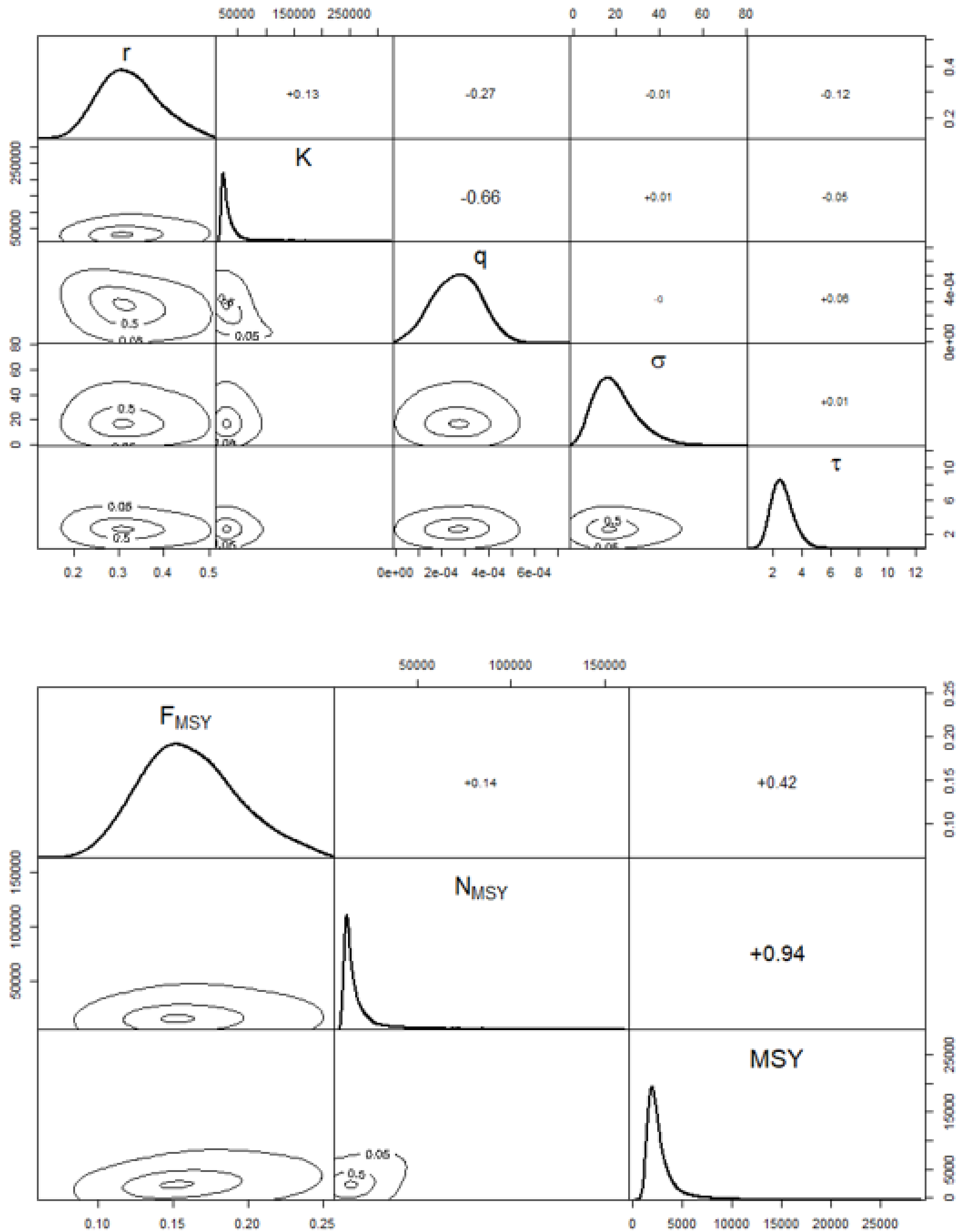


*Figure 24. Probability distributions of kernel model parameters K, r, q, σ, and τ (upper panel) and fisheries management parameters maximum sustainable yield (MSY) fishing mortality ($F_{MSY}$) and abundance at MSY ($N_{MSY}$) (lower panel) from the surplus production model for Rat River Dolly Varden. Isograms (to the left of the diagonal) indicated covariate relations and values in boxes (to the right of the diagonal) are Spearman correlation coefficients. The definition of all model parameters can be found in Table 4.*

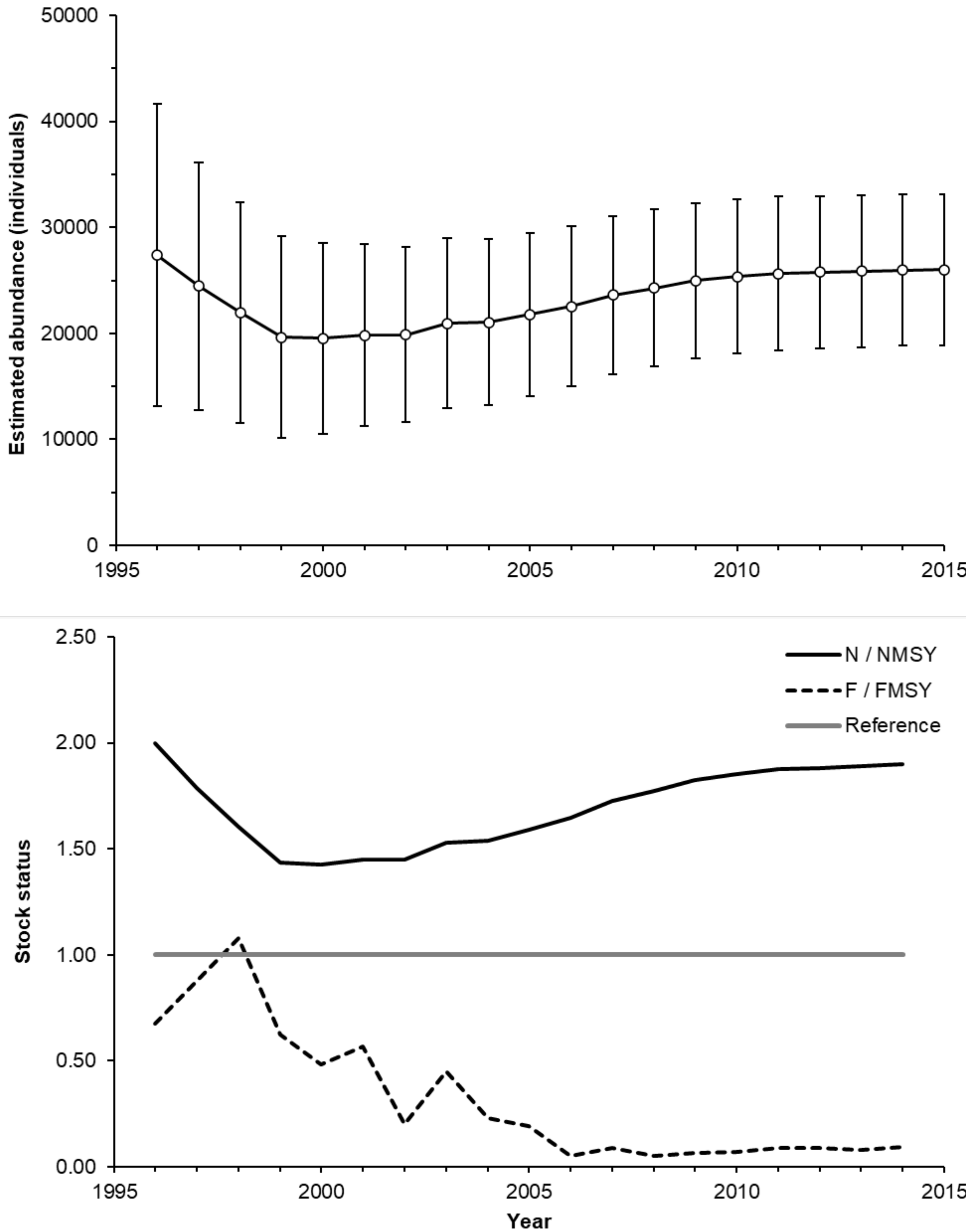


*Figure 25. Population abundance (upper panel) and abundance ($N/N_{MSY}$) and fishing mortality ($F/F_{MSY}$) statuses (lower panel) estimated by surplus production model for Rat River Dolly Varden, 1996–2014. The reference line is $N/N_{MSY}$ or $F/F_{MSY}$ = 1.*

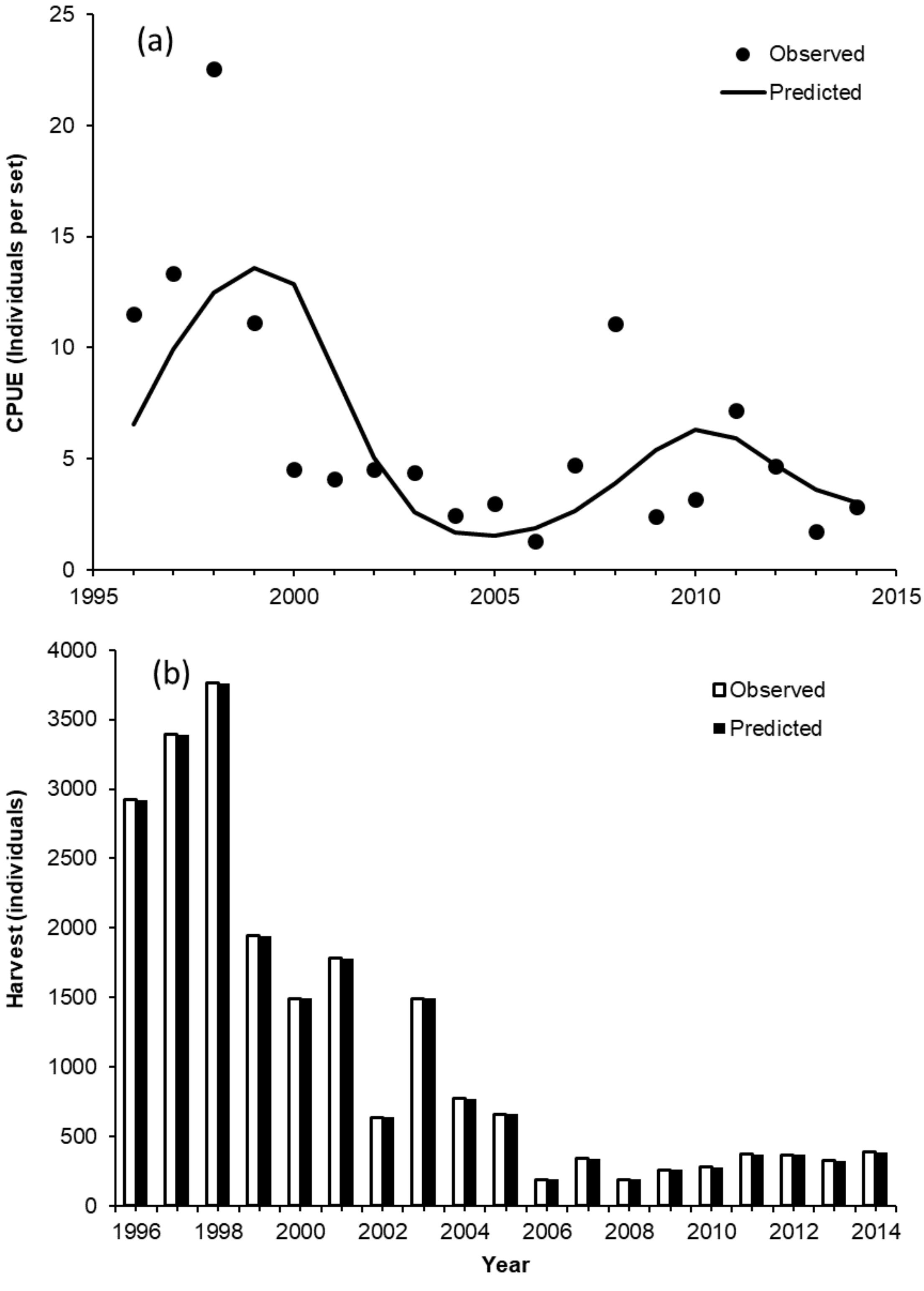


*Figure 26. Standardized and predicted catch-per-unit effort (CPUE) (a) and subsistence harvest (b) from the statistical catch-at-age model for Rat River Dolly Varden,1996–2014.*

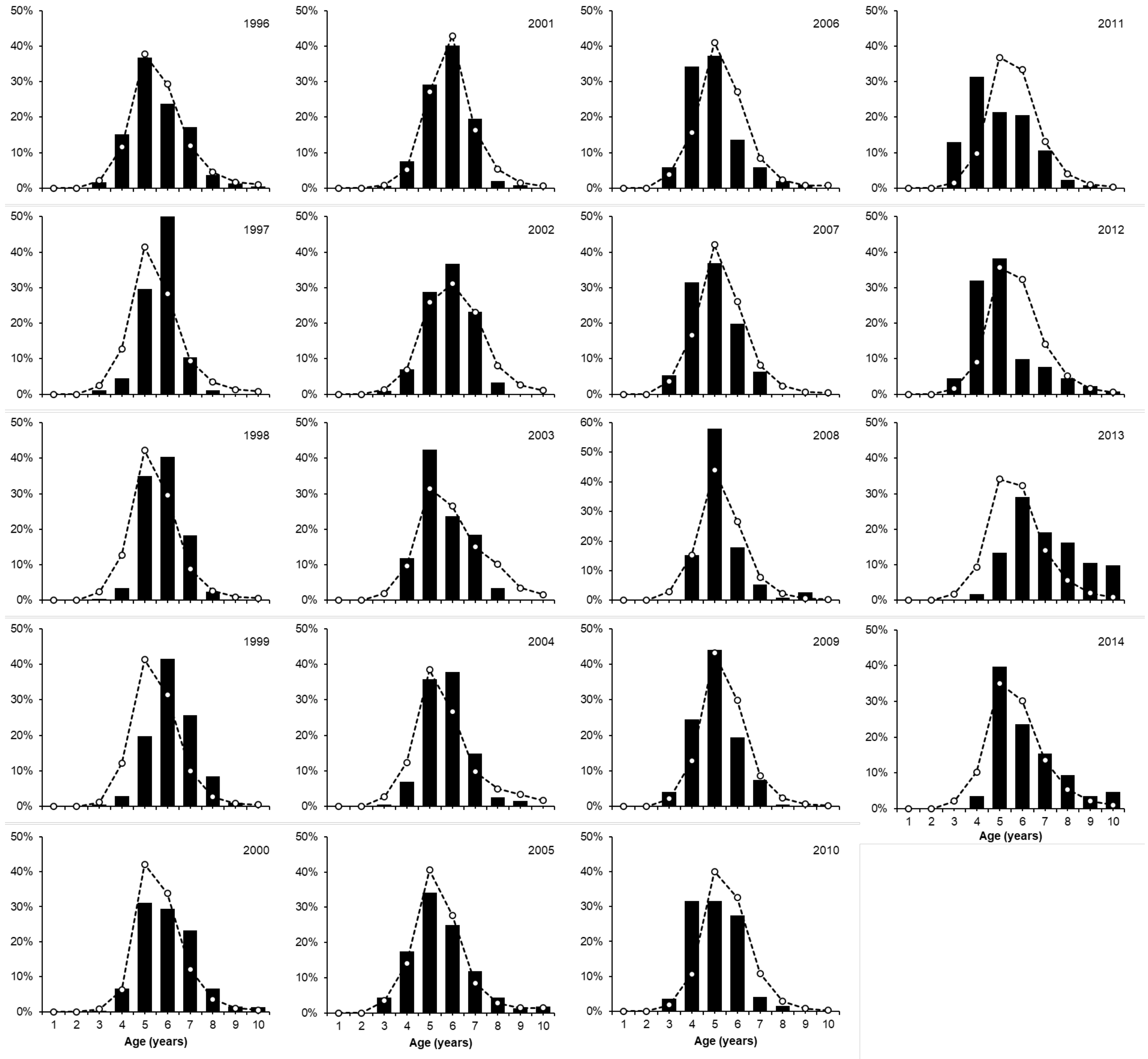


*Figure 27. Observed (black bars) and predicted (open circles and dashed line) age compositions from the statistical catch-at-age model for Rat River Dolly Varden, 1996–2014.*

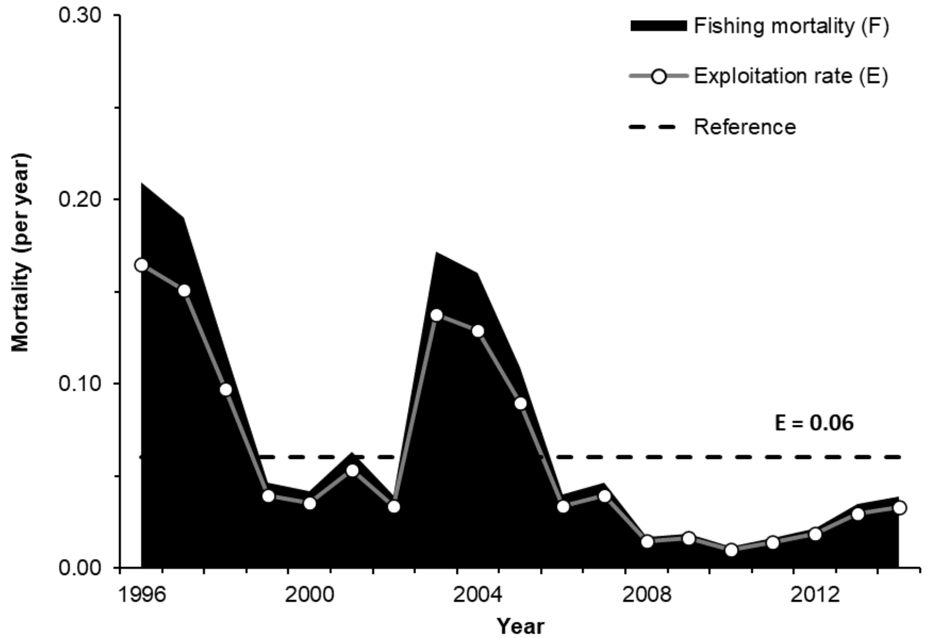


*Figure 28. Fishing mortality and exploitation rate (E) for Rat River Dolly Varden, 1996–2014. Reference represents the average exploitation rate.*

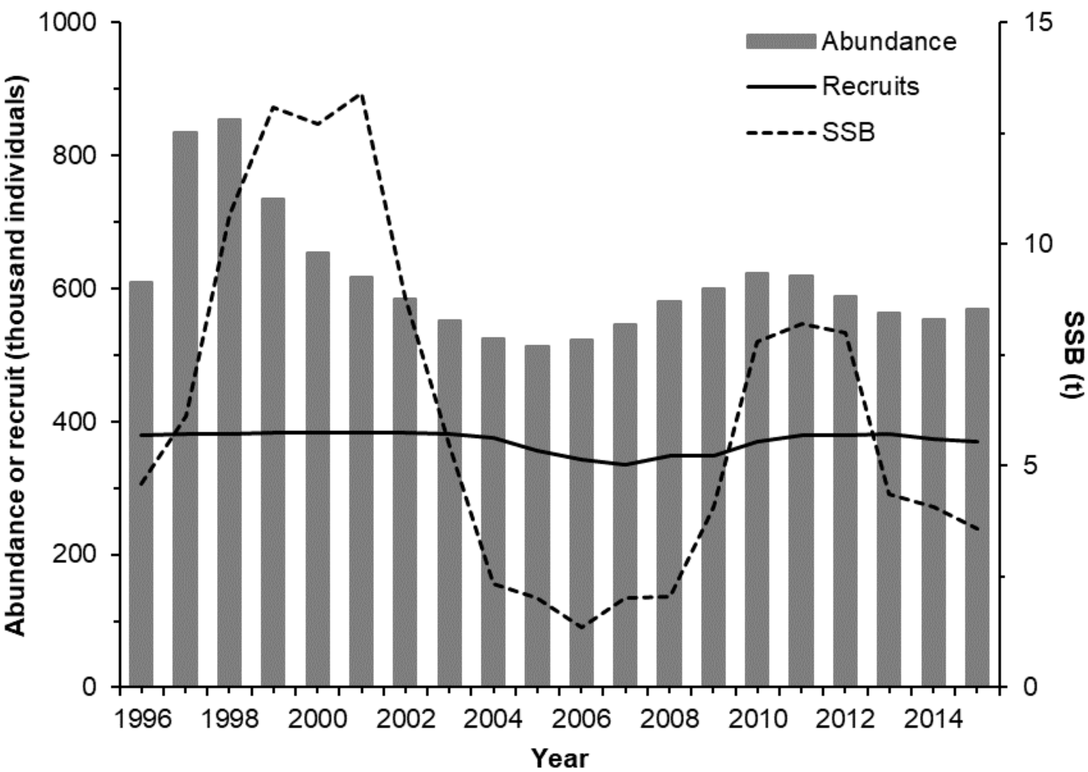


*Figure 29. Total abundance (all age classes), recruits (age 1 abundance) and stock spawning biomass (SSB) of 50% age 6 plus all age 7+ Rat River Dolly Varden, estimated by statistical catch-at-age models for Rat River Dolly Varden, 1996–2014.*

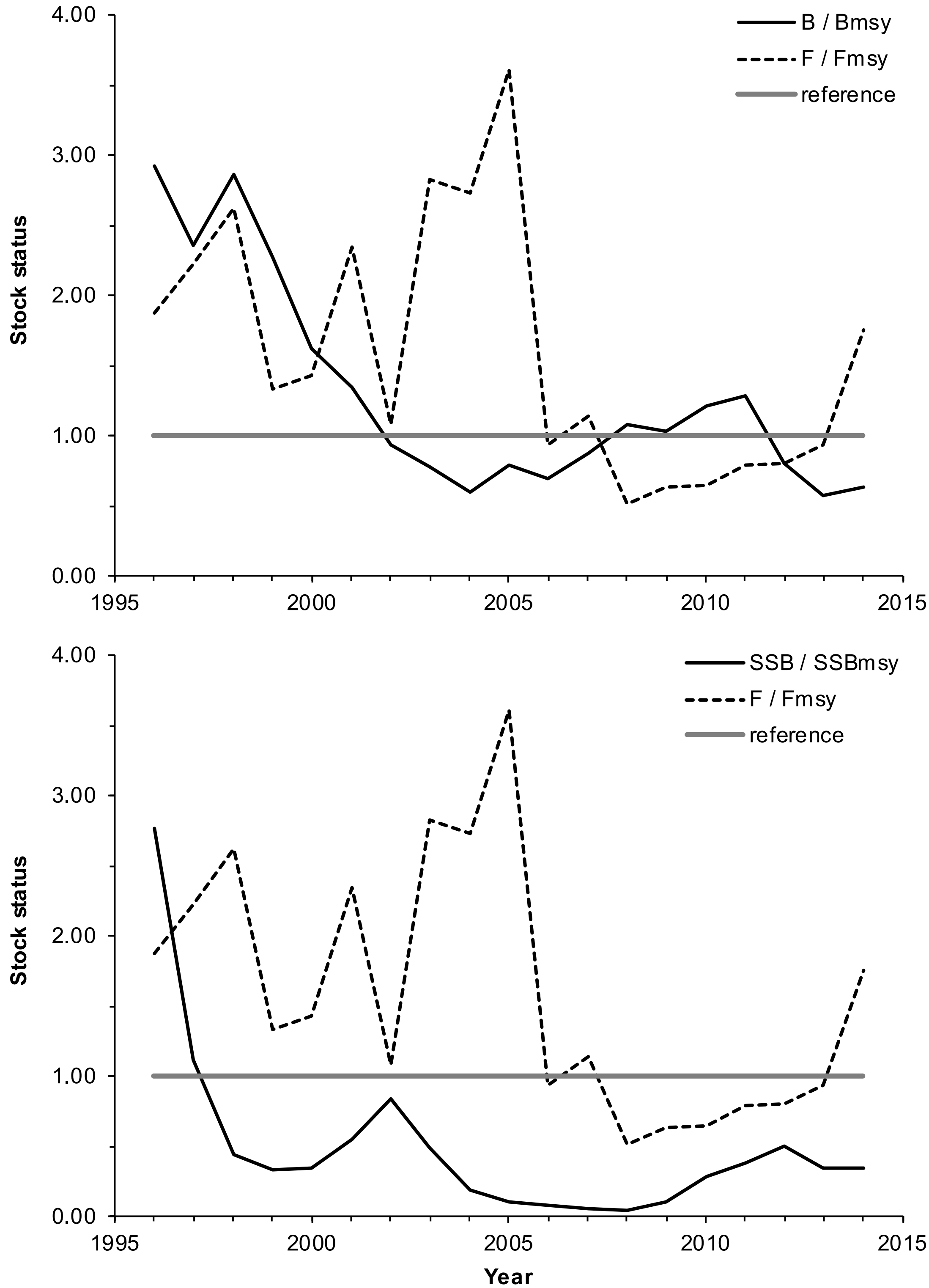


*Figure 30. Total biomass (B) relative to biomass at maximum sustainable yield ($B_{MSY}$) (upper panel) and total spawning stock biomass (SSB) relative to spawning biomass at MSY ($SSB_{MSY}$) (lower panel) compared to removal rates, fishing mortality relative to fishing at MSY ($F/F_{MSY}$) for subsistence fisheries of Rat River Dolly Varden, 1996–2014. The reference line is $B/B_{MSY}$ or $F/F_{MSY}$ = 1.*

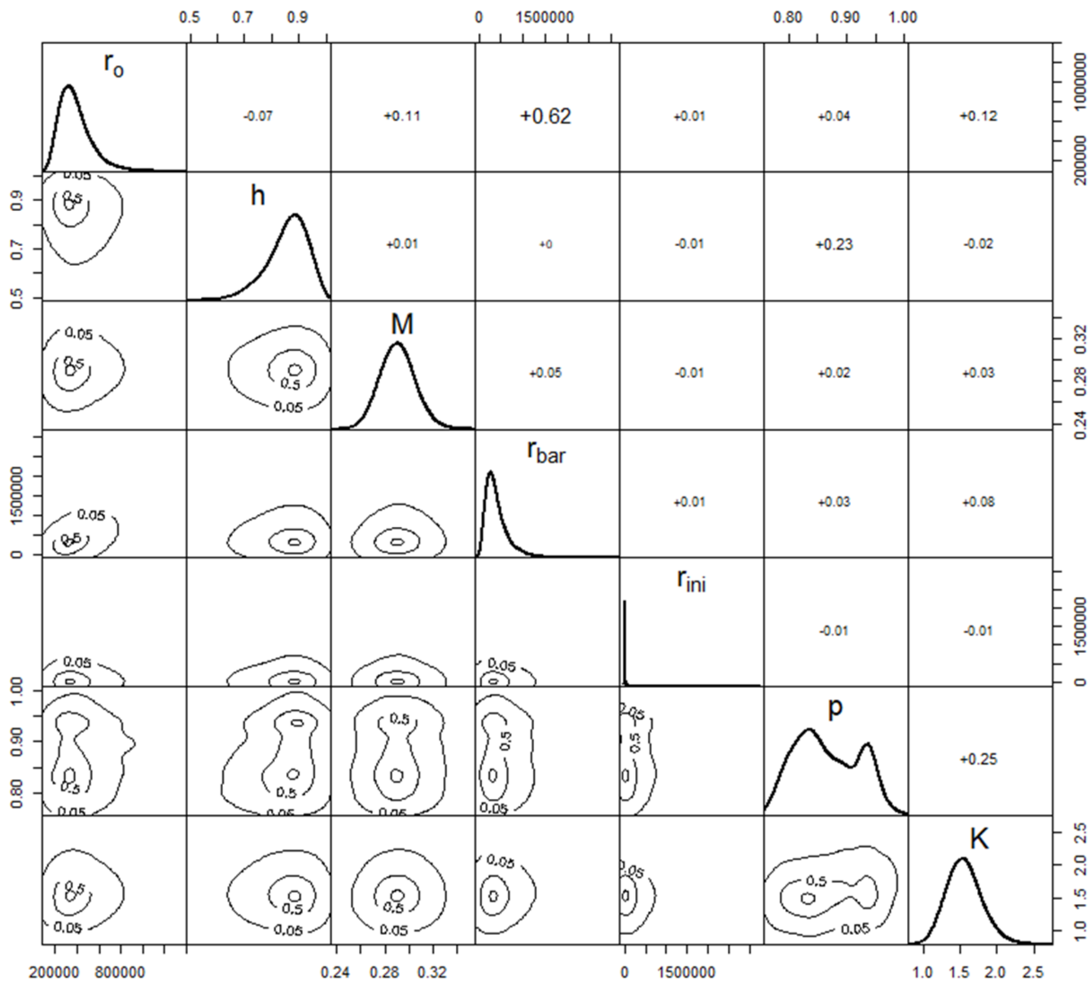


*Figure 31. Probability distribution of statistical catch-at-age parameters including unfished recruits ($r_0$), stock-recruit relationship steepness (h), natural mortality (M), average recruits (rbar), initial value of recruit ($r_{ini}$), ratio of the process to total error (ρ), and recruitment compensation (κ) for Rat River Dolly Varden. Isograms (to left of diagonal) indicated covariate relations and values in boxes (to right of diagonal) are Spearman correlation coefficients.*

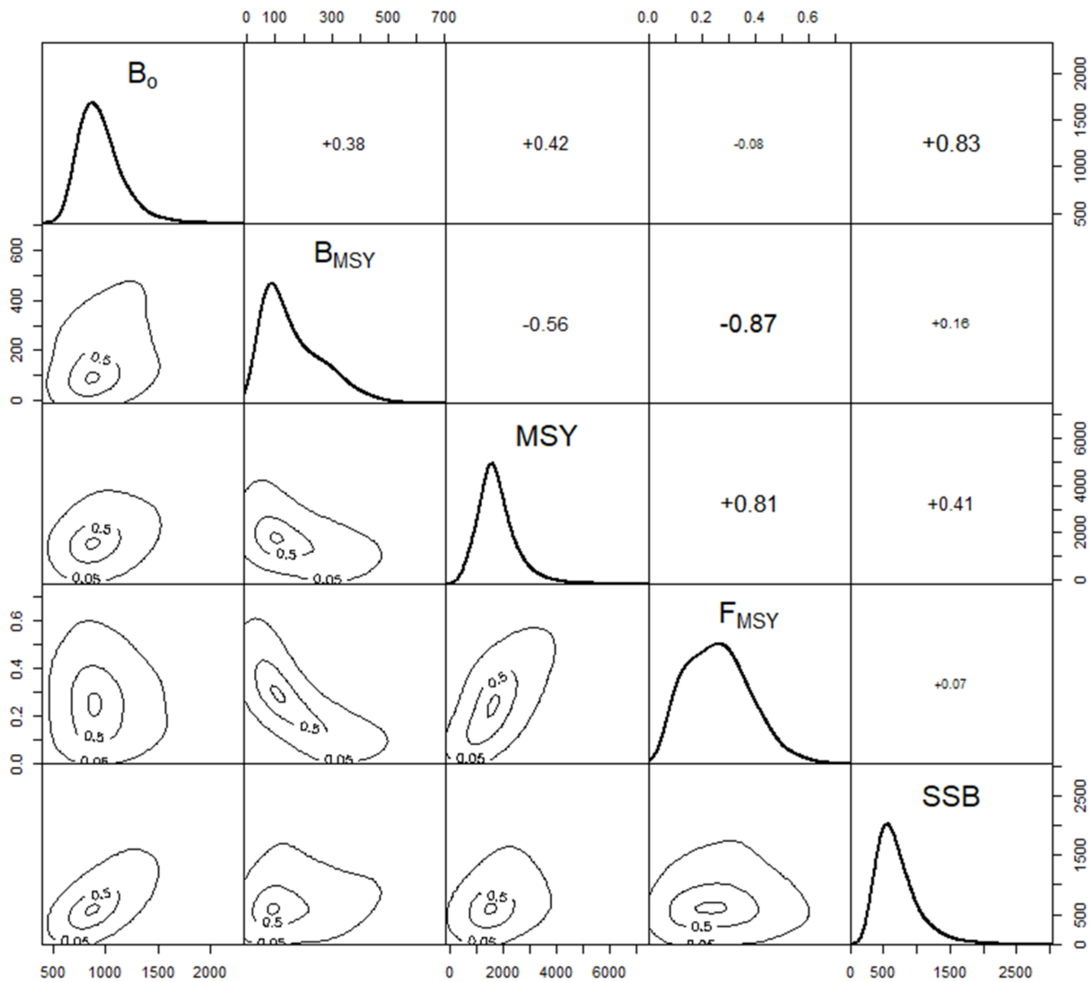


*Figure 32. Probability distribution of virgin biomass ($B_0$), maximum sustainable yield (MSY), biomass and fishing mortality at MSY ($B_{MSY}$ and $F_{MSY}$), and spawning stock biomass (SSB) for Rat River Dolly Varden. Isograms (to left of diagonal) indicated covariate relations and values in boxes (to the right of diagonal) are Spearman correlation coefficients.*